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Information on the species and habitats around the coasts and sea of the British Isles

Mediomastus fragilis and cirratulids in infralittoral mixed sediment

MarLIN – Marine Life Information Network
Marine Evidence–based Sensitivity Assessment (MarESA) Review

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- Core records
- Non-core, certain determination
- Non-core, uncertain determination
- ▢ Predicted habitat extent

Biotope distribution data provided by the JNCC (mhc.jncc.gov.uk)

Researched by	Dr Harvey Tyler-Walters, Kelsey Lloyd & Amy Watson	Refereed by	This information is not refereed

Summary

☰ UK and Ireland classification

EUNIS 2008	A5.435TMP	*Mediomastus fragilis* and cirratulids in infralittoral mixed sediment
JNCC 2022	SS.SMx.IMx.MedCirr	*Mediomastus fragilis* and cirratulids in infralittoral mixed sediment
JNCC 2015	None	None
JNCC 2004	None	None
1997 Biotope	None	None

🏷 Description

Infralittoral shallow mixed sediment, characterised by a diverse number of cirratulid polychaetes, bivalves and amphipods. This biotope has been found in Bembridge (recommended MCZ), located in the Eastern English Channel adjacent to the eastern end of the Isle of Wight. The description of this biotope is based on infauna recorded from the above location but could be found in other areas with similar environmental conditions. The most characterizing species include *Mediomastus fragilis* and a wider cirratulid genera i.e.*Chaetozone*, *Aphelochaeta*, *Caulleriella* and *Cirrfiromia* often with nuculid bivalves *Nucula nucleus* and *Melinna palmata*. The other polychaetes include *Lumbrineris aniara*, *Nephtys kersivalensis*, *Galathowenia oculata* followed by amphipods *Harpinia* and *Ampelisca*. This biotope was described using Day grab infaunal data and the characterizing species listed will partly reflect the method

used to collect data. (Information from JNCC, 2022).

↓ Depth range

5-10 m

🏛 Additional information

-

✔ Listed By

- none -

⌗ Further information sources

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Sensitivity review

Sensitivity characteristics of the habitat and relevant characteristic species

This mixed sediment biotope (SS.SMx.IMx.MedCirr) is characterized by circalittoral gravelly muddy sand and muddy sandy gravels. It is probably similar to SS.SCS.CCS.MedLumVen and SS.SCS.ICS.MoeVen biotopes found in coarse sediment, but the community differs due to the increased mud component and the resultant increase in cirratulids. The most characterizing species include *Mediomastus fragilis* and cirratulid genera i.e. *Chaetozone, Aphelochaeta, Caulleriella* and *Cirrfiromia* often with nuculid bivalves *Nucula nucleus* and *Melinna palmata*. The other polychaetes include *Lumbrineris aniara, Nephtys kersivalensis, Galathowenia oculata* followed by amphipods *Harpinia* and *Ampelisca* (JNCC, 2022). The sediment and hydrodynamics are considered to be key physical factors structuring the biotope and their sensitivity is, therefore, considered for pressures that may lead to alterations. The dominant polychaetes are considered the key characterizing species and the sensitivity assessments focus on these together with *Nucula* spp., while evidence for the other bivalves and species are considered generally.

Resilience and recovery rates of habitat

A large number of species are recorded in the biotope and there may be large natural variation in species abundance over the course of a year or between years. These variations may not alter the biotope classification where habitat parameters, such as sediment type, remain as described in the classification and many of the characteristic species groups are present. For many of the bivalve species studied, recruitment is sporadic and depends on a successful spat fall but recruitment by the characterizing polychaetes may be more reliable. However, due to the large number of pre and post-recruitment factors such as food supply, predation, and competition, the recruitment of venerid bivalves and other species is unpredictable (Olafsson *et al.*, 1994). The species that are present in the biotope can be broadly characterized as either opportunist species that rapidly colonize disturbed habitats and increase in abundance, or species that are larger and longer-lived and that may be more abundant in an established, mature assemblage.

Species with opportunistic life strategies (small size, rapid maturation and short lifespan of 1-2 years with production of large numbers of small propagules), include the characterizing polychaetes *Mediomastus fragilis*. These are likely to recolonize disturbed areas first, although the actual pattern will depend on the recovery of the habitat, season of occurrence and other factors. MES (2010) reported that *Mediomastus* sp. produced large numbers of small eggs, fertilized externally and deposited as egg masses of ca 500 eggs per individual. They hatch into planktotrophic larvae that spend ca four weeks in the water column. MES (2010) concluded that a large number of eggs and planktonic dispersal suggested a moderate to high recovery potential, although they also noted that little was known about its life history.

Nephtys sp. is a relatively long-lived polychaete with a lifespan of six to possibly as much as nine years. It matures at one year and the females release over 10,000 (and up to 80,000 depending on species) eggs of 0.11-0.12 mm from April through to March. These are fertilized externally and develop into an early lecithotrophic larva and a later planktotrophic larva which spends as much as 12 months in the water column before settling from July to September. The genus *Nephtys* has a relatively high reproductive capacity and widespread dispersion during the lengthy larval phase. It is likely to have a high recoverability following disturbance (MES, 2010).

Many cirratulids are thought to have direct development so adult dispersal is likely to be low, especially if the adults are also sedentary burrow dwellers, e.g. *Chaetozone* sp. However, MES (2010) reported that *Chaetozone* sp. has a short lifespan (1-2 years), became sexually mature in less than one year, and had planktonic dispersal and rapid growth rate, and suggested it was opportunistic. Conversely, *Caulleriella* sp. was thought to have a lifespan of 3-5 years, and did not reach maturity until its second year. It produces a large number of eggs (1000-5000). But it was not known if the larvae are planktonic or are brooded, so dispersal would be low. MES (2010) concluded that, while local recruitment is probably high, recolonization from adjacent areas or distant populations may be prolonged and recovery of its abundance could take 3-5

years.

George (1968) discussed possible recolonization in the two cirratulids *Cirratulus cirratus* and *Cirriformia tentaculata* in the British Isles. Following the disappearance of this species from Sussex after the severe winter of 1962-63, he suggested that *Cirratulus cirratus* probably existed subtidally in such small numbers that it could not maintain itself once replenishment from the shore population had ceased. It was concluded that recolonization by *Cirriformia tentaculata* would be by marginal dispersal rather than remote dispersal (Crisp, 1958, cited in George, 1968) and that it was likely to take several decades with mild winters before its distribution returned to that prior to 1962/63 (George, 1968).

The lifecycle of *Aphelochaeta marioni* varies according to environmental conditions. In Stonehouse Pool, Plymouth Sound, *Aphelochaeta marioni* (studied as *Tharyx marioni*) spawned in October and November (Gibbs, 1971) whereas, in the Wadden Sea, Netherlands, spawning occurred from May to July (Farke, 1979). Spawning, which occurs at night, was observed in a microsystem in the laboratory by Farke (1979). The female rose up into the water column with the tail end remaining in the burrow. The eggs were shed within a few seconds and sank to form puddles on the sediment. The female then returned to the burrow and resumed feeding within half an hour. Fertilization was not observed, probably because the male does not leave the burrow. The embryos developed lecithotrophically and hatched in about 10 days (Farke, 1979). The juveniles dug into the sediment immediately after hatching. Where the sediment depth was not sufficient for digging, the juveniles swam or crawled in search of a suitable substratum (Farke, 1979). In the microsystem, juvenile mortality was high (ca 10% per month) and most animals survived for less than a year (Farke, 1979). In the Wadden Sea, the majority of the cohort reached maturity and spawned at the end of their first year, although some slower developers did not spawn until the end of their second year (Farke, 1979). However, the population of *Aphelochaeta marioni* in Stonehouse Pool spawned for the first time at the end of the second year of life (Gibbs, 1971). There was no evidence of major post-spawning mortality and it was suggested that individuals may survive to spawn over several years. Gibbs (1971) found that the number of eggs laid varied from 24-539 (mean=197) and was correlated with the female's number of genital segments, and hence, female size and age. Therefore, if adjacent populations are available recovery will be rapid but where the affected population is isolated or severely reduced, recovery may be extended. However, Farke (1979) implied that *Aphelochaeta marioni* (studied as *Tharyx marioni*) became dominant in areas of the German Bight, where it was previously absent, in only a few years. However, the recoverability of cirratulids as a group is likely to be low.

In *Nucula nitidosa* from the German Bight, the timing of spawning in the summer and autumn was attributed to the seasonal rise in temperature during the summer months. However, at Plymouth, the same species appeared to breed in winter when bottom temperatures are falling, as is the case in *Pronucula tenuis* from Loch Etive (Harvey & Gage, 1995). The availability of a suitable food supply during the months prior to spawning may be a more potent determinant of spawning time (Berry, 1989; Tyler *et al.*, 1992, both cited in Harvey & Gage, 1995), with annual variation in the availability and quality of food determining the exact time of spawning in any one year. The lifespan of *Nucula nitidosa* ranges from 7-10 years (Wilson, 1992). It takes 2-3 years for *Nucula nitidosa* to reach sexual maturity (Davis & Wilson, 1983b) and reproduce in high numbers. Once hatched, *Nucula nitidosa* larvae spend a short time in the water column (a few days), which reduces the risk of predation. However, juveniles do not have a high dispersal potential as they settle in the vicinity of the adults (Thorson, 1946). Populations of *Nucula nitidosa* appear stable and were reported to fluctuate little from year to year (Thorson, 1946). Rachor (1976) reported that the mortality rate of *Nucula nitidosa* was very uncertain. Populations of *Nucula nitidosa* can increase markedly when the bottom sediments are suitable and decrease when the older age classes die. For instance, in Dublin Bay, low larval and adult mortality rates of *Nucula nitidosa* were reported for several years, which was followed by high mortality when adults reached old age (Davis & Wilson, 1983b). *Nucula nitidosa* is also known to inhabit unstable substrata and populations can reach high densities (Creutzberg, 1986). Hence, *Nucula nitidosa* is likely to exhibit good local recruitment. Therefore, if the extent or abundance of a population is reduced, recovery is likely to be rapid. However, long-distance dispersal is potentially poor. If a population is removed from an area, it may take a long time for the area to be recolonized, depending on the local hydrography.

The amphipod genus *Ampelisca* has some life history traits that allow them to recover quickly where populations are disturbed. They do not produce large numbers of offspring but reproduce regularly and the larvae are brooded, giving them a higher chance of survival within a suitable habitat than free-living larvae. *Ampelisca* has a short lifespan and reaches sexual maturity in a matter of months allowing a population to recover abundance and biomass in a very short period of time (MES, 2010). Experimental studies have shown *Ampelisca abdita* to be an early colonizer, in large abundances of defaunated sediments where local populations exist to support recovery (McCall, 1977) and *Ampelisca abdita* have been shown to migrate to, or from, areas to avoid unfavourable conditions (Nichols & Thompson, 1985). *Ampelisca* spp. are very intolerant of oil contamination and the recovery of the *Ampelisca* populations in the fine sand community in the Bay of Morlaix took up to 15 years following the *Amoco Cadiz* oil spill, probably due to the amphipods' low fecundity, lack of pelagic larvae and the absence of local unperturbed source populations (Poggiale & Dauvin, 2001).

A number of studies have tracked the recovery of sand and coarse sand communities following disturbance from fisheries (Gilkinson *et al.*, 2005) and aggregate extraction (Boyd *et al.*, 2005). The available studies confirm the general trend that, following severe disturbance, habitats are recolonized rapidly by opportunistic species (Pearson & Rosenberg, 1978). Experimental deployment of hydraulic clam dredges on a sandy seabed on Banquereau, on the Scotian Shelf, eastern Canada showed that within two years of the impact, polychaetes and amphipods had increased in abundance after one year (Gilklinson *et al.*, 2005). Two years after dredging, abundances of opportunistic species were generally elevated relative to pre-dredging levels while communities had become numerically dominated (50-70%) by *Spiophanes bombyx* (Gilkinson *et al.*, 2005). Van Dalfsen *et al.* (2000) found that polychaetes recolonized a dredged area within 5-10 months (cited by Boyd *et al.*, 2005), with biomass recovery predicted within 2-4 years. Therefore, the polychaete and amphipods are therefore likely to recover more rapidly than the characteristic bivalves.

In an area that had been subjected to intensive aggregate extraction for 30 years, the abundance of juvenile and adults *Nephtys cirrosa* had greatly increased three years after extraction had stopped (Mouleaert & Hostens, 2007). An area of sand and gravel subject to chronic working for 25 years had not recovered after six years when compared to nearby reference sites unimpacted by operations (Boyd *et al.*, 2005). Sardá *et al.* (1999) tracked annual cycles within a *Spisula* community in the Bay of Blanes (northwest Mediterranean sea, Spain) for 4 years. Macroinfaunal abundance peaked in spring, and decreased sharply throughout the summer, with low density in autumn and winter. The observed trends were related to a number of species, including many that characterize this biotope such as *Mediomastus fragilis*. *Mediomastus fragilis* had spring population peaks but more individuals persisted throughout the year.

Where impacts also alter the sedimentary habitat, recovery of the biotope will also depend on the recovery of the habitat to the former condition to support the characteristic biological assemblage. Recovery of sediments will be site-specific and will be influenced by currents, wave action and sediment availability (Desprez, 2000). Except in areas of mobile sands, the process tends to be slow (Kenny & Rees, 1996; Desprez, 2000). Boyd *et al.* (2005) found that in a site subject to long-term extraction (25 years), extraction scars were still visible after six years and sediment characteristics were still altered in comparison with reference areas, with ongoing effects on the biota.

Resilience assessment. Where resistance is 'None' or 'Low' and an element of habitat recovery is required, resilience is assessed as **'Medium'** (2-10 years), based on evidence from aggregate recovery studies in similar habitats including Boyd *et al.* (2005). Where resistance of the characterizing species is 'Low' or 'Medium' and the habitat has not been altered, resilience is assessed as **'High'** as, due to the number of characterizing species and variability in recruitment patterns, it is likely that the biotope would be considered representative and hence recovered after two years although some parameters such as species richness, abundance and biotopes may be altered. Recovery of the seabed from severe physical disturbances that alter sediment character may also take up to 10 years or longer (Le Bot *et al.*, 2010), although extraction of gravel may result in more permanent changes and this will delay recovery.

Note: The resilience and the ability to recover from human-induced pressures is a combination of the environmental conditions of the site, the frequency (repeated disturbances versus a one-off event) and the

intensity of the disturbance. Recovery of impacted populations will always be mediated by stochastic events and processes acting over different scales including, but not limited to, local habitat conditions, further impacts and processes such as larval supply and recruitment between populations. Full recovery is defined as the return to the state of the habitat that existed prior to impact. This does not necessarily mean that every component species has returned to its prior condition, abundance or extent but that the relevant functional components are present and the habitat is structurally and functionally recognizable as the initial habitat of interest. It should be noted that the recovery rates are only indicative of the recovery potential.

🚢 Hydrological Pressures

	Resistance	Resilience	Sensitivity
Temperature increase (local)	High	High	Not sensitive
	Q: Medium A: Medium C: Medium	Q: High A: High C: High	Q: Medium A: Medium C: Medium

Little direct evidence was found to support the assessment of this pressure. Few laboratory studies have been carried out and the sensitivity assessment is based on studies monitoring settlement and recruitment and records of species distribution.

Kröncke *et al.* (1998) examined long-term changes in the macrofauna in the subtidal zone off Norderney, one of the East Frisian barrier islands. The analysis suggested that macrofauna were severely affected by cold winters whereas storms and hot summers have no impact on the benthos. A long-term increase in temperature might cause a shift in species composition. Long-term analysis of the North Sea pelagic system has identified yearly variations in larval abundance of Echinodermata, Arthropoda, and Mollusca larvae that correlate with sea surface temperatures. Larvae of benthic echinoderms and decapod crustaceans increased after the mid-1980s, coincident with a rise in North Sea sea surface temperature, whereas bivalve larvae underwent a reduction (Kirby *et al.*, 2008). An increase in temperature may alter larval supply and, in the long-term and over large spatial scales, may result in changes in community composition.

Mediomastus fragilis is recorded from northern Norway, south to the Iberian Peninsula and into the Mediterranean, and across the North Sea into the Baltic (OBIS, 2022). OBIS records reported a sea surface temperature range of -5°C to 25°C but the majority of records were in the range of 10-15°C (OBIS, 2022). *Chaetozone gibber* has a similar distribution from Norway, south to the Iberian Peninsula and into the Mediterranean and is recorded from a sea surface temperature of 5-20°C, although most records are at 10-15°C. *Nucula nucleus* is recorded from northern Norway, south to the Iberian Peninsula and into the Mediterranean but is also recorded in South Africa (OBIS, 2022). OBIS records reported a sea surface temperature range of 0°C to 25°C but the majority of records were in the range of 10-15°C (OBIS, 2022). The cirratulid *Aphelochaeta marioni* (studied as *Tharyx marioni*) has been recorded from the Baltic to the Indian Ocean and so it probably has some degree of adaptation or tolerance to a range of temperatures (Hartmann-Schroder, 1974; Rogall, 1977, cited in Farke, 1979). However, acute rises in temperature may have a more deleterious effect.

George (1964a) reported that a rapid rise or fall in temperature of 3°C was sufficient to induce spawning in 25% of mature *Cirriformia tentaculata*. If this occurred at a time of year that was not suitable for larval survival then larval mortality could be high. The upper lethal limits for *Cirriformia tentaculata* from the Hamble were reported to be 32°C and 29°C for 5-6 day old and adult *Cirriformia tentaculata* respectively (George, 1964b). *Cirriformia tentaculata* is reported to be near its northern limit in the British Isles (George, 1968) and an increase in temperature may lead to the extension of its upper distribution range. An increase in temperature could also serve to decrease the length of time spent in the larval phase and so reduce the risk of predation. The rate of larval growth in *Cirriformia tentaculata* was found to be twice as fast at 20°C as at 8°C.

Sensitivity assessment. Little direct evidence was available to assess this pressure. Most of the characteristic species occur to the north or south of UK waters so it is likely that they would be resistant to an increase in temperature at the benchmark level. An acute change may lead to spawning or other

sublethal biological effects. Therefore, resistance is assessed as **'High'** at the benchmark level. Hence, resilience is 'High' and sensitivity is assessed as **'Not sensitive' at the benchmark level.**

Temperature decrease (local)	Medium	High	Low
	Q: High A: Medium C: Medium	Q: High A: Medium C: Medium	Q: High A: Medium C: Medium

Little direct evidence was found to support the assessment of this pressure. Few laboratory studies have been carried out and the sensitivity assessment is based on studies monitoring settlement and recruitment and records of species distribution. Kröncke *et al*. (1998) examined long-term changes in the macrofauna in the subtidal zone off Norderney, one of the East Frisian barrier islands. The analysis suggested that macrofauna were severely affected by cold winters whereas storms and hot summers have no impact on the benthos. A long-term increase in temperature might cause a shift in species composition.

Mediomastus fragilis is recorded from northern Norway, south to the Iberian Peninsula and into the Mediterranean, and across the North Sea into the Baltic (OBIS, 2022). OBIS records reported a sea surface temperature range of -5°C to 25°C but the majority of records were in the range of 10-15°C (OBIS, 2022). *Chaetozone gibber* has a similar distribution from Norway, south to the Iberian Peninsula and into the Mediterranean and is recorded from a sea surface temperature of 5-20°C, although most records are at 10-15°C. *Nucula nucleus* is recorded from northern Norway, south to the Iberian Peninsula and into the Mediterranean but is also recorded in South Africa (OBIS, 2022). OBIS records reported a sea surface temperature range of 0°C to 25°C but the majority of records were in the range of 10-15°C (OBIS, 2022). The cirratulid *Aphelochaeta marioni* (studied as *Tharyx marioni*) has been recorded from the Baltic to the Indian Ocean and so it probably has some degree of adaptation or tolerance to a range of temperatures (Hartmann-Schroder, 1974; Rogall, 1977, cited in Farke, 1979). However, acute rises in temperature may have a more deleterious effect.

George (1964a) reported that a rapid rise or fall in temperature of 3°C was sufficient to induce spawning in 25% of mature *Cirriformia tentaculata*. If this occurred at a time of year that was not suitable for larval survival then larval mortality could be high. However, George (1964b) noted that although in Southampton the incoming tide incurred a drop of 6 °C in five minutes, such rapid changes in temperature had no significant effect on the mortality of either juvenile or adult *Cirriformia tentaculata* in the laboratory. The larvae of this species grow twice as slow at 8°C as they do at 20°C (George, 1964a). Any increase in the length of time spent in the larval phase will increase the risk of predation. In adults, field data suggests that growth ceases at 6°C (George, 1964a). On the Hamble, lower lethal limits of -6°C (by extrapolation) and 2°C have been reported for 5-6 day old and adult *Cirriformia tentaculata* respectively (George, 1964b). These are temperatures that can reasonably be expected in winter in this intertidal biotope and so some mortality is likely. Furthermore, *Cirriformia tentaculata* is reported to be near its northern limit in the British Isles (George, 1968) and a long-term chronic decrease in temperature could serve to exclude this species from the northern extent of its distribution. George (1968) reported several major changes and a major reduction in the distribution range of *Cirriformia tentaculata* following the severe winter of 1962/3. In temperature tolerance experiments, no *Cirriformia tentaculata* survived even a brief exposure to -2°C or 96 hours at 0°C.

The cirratulid *Cirratulus cirratus* was found to tolerate lower temperatures and it is possible that this species will become more prevalent in this biotope if the temperature falls. George (1968) reported that the ciliary feeding mechanisms of *Cirriformia tentaculata* became so inefficient at low temperatures that, over long periods, the animal may die of starvation. George (1968) also mentioned that the animal does not withdraw its branchiae in cold weather. Due to their delicate nature, the branchiae may subsequently freeze on the surface. In such a case, the animal would be living under anaerobic conditions and so emerges from the burrow to enable them to respire through their body surface.

Long-term analysis of the North Sea pelagic system has identified yearly variations in larval abundance of Echinodermata, Arthropoda, and Mollusca larvae that correlate with sea surface temperatures. Larvae of benthic echinoderms and decapod crustaceans increased after the mid-1980s, coincident with a rise in

North Sea sea surface temperature, whereas bivalve larvae underwent a reduction (Kirby *et al*., 2008). A decrease in temperature may alter larval supply and in the long-term, over large spatial scales, may result in changes in community composition.

Sensitivity assessment. Little direct evidence was available to assess this pressure. Most of the characteristic species occur to the north or south of UK waters so it is likely that they would be resistant to an increase in temperature at the benchmark level. An acute change may exceed thermal tolerances or lead to spawning or other sublethal effects. The exception is *Cirriformia tentaculata*, whose population may be severely reduced by acute changes typical of severe winter temperatures in the UK. . Therefore, resistance is assessed as **'Medium'**. Hence, resilience is assessed as **'High'** and sensitivity as **'Low' at the benchmark level.**

Salinity increase (local)	Medium	High	Low
	Q: Low A: NR C: NR	Q: High A: Medium C: Medium	Q: Low A: Low C: Low

No directly relevant evidence was found to assess this pressure. A study from the Canary Islands indicated that exposure to high salinity effluents (47- 50 psu) from desalination plants altered the structure of biological assemblages, reducing species richness and abundance (Riera *et al*., 2012). Bivalves and amphipods appeared to be less tolerant of increased salinity than polychaetes and were largely absent at the point of discharge. Polychaetes, including species or genera that occur in this biotope, such as *Lumbrineris* spp. were present at the discharge point (Riera *et al*., 2012). However, Russo *et al*. (2006, 2007; cited in Roberts et al., 2010b) reported that polychaete abundance and diversity decreased in proximity to the hypersaline discharge, and noted that sensitivity varied between polychaetes families with Ampharetidae being the most sensitive. Roberts *et al*. (2010b) noted that the effects of desalination plants effluents varied from none to significant impacts in seagrass, coral reef and soft-sediment communities in poorly flushed environments. They also noted that in most other cases the effects were limited to within 10s of metres of the outfalls.

Sensitivity assessment. High saline effluents have the potential to alter the structure of biological assemblages in close proximity to outfalls and/or in poorly flushed locations. Polychaete species may be more tolerant than bivalves but an increase in salinity is likely to result in declines in species richness and abundance based on Roberts *et al*. (2010b) and Riera *et al*. (2012). This biotope (SS.SMx.IMx.MedCirr) is found in full to variable salinity waters, moderately exposed to sheltered from wave action and in strong to weak tidal streams. It is probably well flushed so that exposure to hypersaline conditions may be limited to the close proximity to any outfall. Therefore, resistance is assessed as **'Medium'**. Hence, resilience is assessed as **'High'** and sensitivity is assessed as **'Low'**, albeit with 'Low' confidence due to the lack of direct evidence.

Salinity decrease (local)	Low	High	Low
	Q: Low A: NR C: NR	Q: High A: Medium C: Medium	Q: Low A: Low C: Low

The polychaetes *Mediomastus fragilis, Chaetozone zetlandica, Aphelochaeta marioni, Caulleriella alata, Cirriformia tentaculata* and the bivalve *Nucula nucleus* and *Melinna palmata* were recorded in areas of 30-35 psu, with few records at lower salinities (OBIS, 2022). Ranchor (1976) successfully reared specimens of *Nucula nitidosa* in the laboratory at a salinity of 27 to 32 ppt. However, very little information on its salinity tolerance was found.

Populations of *Aphelochaeta marioni* inhabit the open coast where seawater is at full salinity. Farke (1979) studied the effects of changing salinity on *Aphelochaeta marioni* (studied as *Tharyx marioni*) in a microsystem in the laboratory. Over several weeks, the salinity in the microsystem was increased from 25-40 psu and no adverse reaction was noted. However, when individuals were removed from the sediment and displaced to a new habitat, they only dug into their new substratum if the salinities in the two habitats were similar. If the salinities differed by 3-5 psu, the worms carried out random digging movements, failed to penetrate the

sediment and died at the substratum surface after a few hours. This would suggest that *Aphelochaeta marioni* can tolerate salinity changes when living infaunally but is less tolerant when removed from its habitat.

Sensitivity assessment. This biotope (SS.SMx.IMx.MedCirr) is found in full to variable (18-35) salinity waters, which suggests some tolerance to a reduction in salinity. A reduction in salinity may result in changes in biotope composition as some sensitive species are lost and replaced by typical estuarine species more tolerant of the changed conditions. Therefore, a reduction in salinity regime from full to reduced (18-35) for a year (the benchmark) may result in a reduction in the abundance of some of the characteristic species in the short term and resistance is assessed as **'Low'.** Hence, resilience is assessed as **'High'** and sensitivity as **'Low' at the benchmark level.**

Water flow (tidal current) changes (local)	High	High	Not sensitive
	Q: Low A: NR C: NR	Q: High A: High C: High	Q: Low A: Low C: Low

This biotope (SS.SMx.IMx.MedCirr) is found in strong (1.5 -3 m/s) and weak (>0.5 m/s) tidal streams and in moderately exposed, sheltered and very sheltered wave conditions (JNCC, 2022). Sands are less cohesive than mud sediments and coarse sediments dominate areas of higher water flow. Hjulström (1939) concluded that fine sand (particle diameter of 0.3-0.6 mm) was easiest to erode and required a mean velocity of 0.2 m/s. Erosion and deposition of particles greater than 0.5 mm require a velocity >0.2 m/s to alter the habitat and gravel (ca 10 mm dia.) may require >1 m/s to erode. The presence of mud may help to consolidate the coarse sediment in areas of strong water flow.

Many of the species occur in a range of sediment types, which, given the link between hydrodynamics and sediment type, suggests that these species are not sensitive to changes in water flow at the pressure benchmark. For example, *Mediomastus fragilis, Chaetozone gibba, Cirriformia tentaculata, Aphelochaeta marioni, Caulleriella zetlandica* and *Nucula nucleus* are found in coarse sediments, muddy sands and sandy muds as well as mixed sediments (JNCC, 2015)

Sensitivity assessment. Changes in water flow may alter the topography of the habitat and may cause some shifts in abundance. An increase to very strong tidal streams may winnow away the biotope while a decrease to weak tidal streams in the absence of wave action may result in the deposition of muddier sediments. However, a change at the pressure benchmark (increase or decrease) is unlikely to affect biotopes that occur in strong to weak flows. Therefore, resistance is assessed as **'High'**, resilience as **'High'**, and sensitivity assessed as **'Not sensitive' at the benchmark level.**

Emergence regime changes	Not relevant (NR)	Not relevant (NR)	Not relevant (NR)
	Q: NR A: NR C: NR	Q: NR A: NR C: NR	Q: NR A: NR C: NR

Changes in emergence are **'Not relevant'** to this biotope which is restricted to fully subtidal habitats.

Wave exposure changes (local)	High	High	Not sensitive
	Q: Low A: NR C: NR	Q: High A: High C: High	Q: Low A: Low C: Low

This biotope (SS.SMx.IMx.MedCirr) is found in strong (1.5 -3 m/s) and weak (>0.5 m/s) tidal streams and in moderately exposed, sheltered and very sheltered wave conditions at a depth of 5-10 metres (JNCC, 2022). It is not directly exposed to the action of breaking waves but to the resultant oscillatory flow at the surface of the sediment. No specific evidence was found to assess this pressure. An increase in wave exposure (however unlikely) to 'exposed' or higher may remove or re-sort the substratum, resulting in coarser or mobile sediments and a potential change in the biotope. A decrease to extremely wave-sheltered conditions is unlikely to result in a significant change in the water flow that characterizes the biotope. In particular, a 3-5% change in significant wave height (the benchmark) is unlikely to result in a significant change. Therefore, resistance is assessed as **'High',** resilience as **'High'** and sensitivity assessed as **'Not**

sensitive' at the benchmark level.

🧪 Chemical Pressures

	Resistance	Resilience	Sensitivity
Transition elements & organo-metal contamination	Medium	High	Low
	Q: Low A: NR C: NR	Q: High A: Medium C: Medium	Q: Low A: Low C: Low

Bryan (1984) suggested that polychaetes were "fairly resistant" to heavy metals. In the short-term, mercury, copper and silver were the most toxic, aluminium, chromium, zinc and lead less toxic and cadmium, nickel, cobalt and selenium the least toxic. The most rapidly absorbed metals (mercury, copper, silver) are generally the most toxic but for others, toxicity was variable and species-specific (Bryan, 1984).

Owenia fusiformis from the south coast of England were found to have loadings of 1335 μg Cu per gram body weight and 784 μg Zn per gram body weight. The metals were bound in spherules within the cells of the gut (Gibbs *et al.*, 2000). No mention was made of any ill effects of these concentrations of metal within the body and it is presumed that *Owenia fusiformis* is tolerant of heavy metal contamination. Rygg (1985) classified *Lumbrineris* spp. as non-tolerant of Cu because the species was only occasionally found at stations in Norwegian fjords where Cu concentrations were >200 ppm (mg/kg).

In Restronguet Creek, the sediments contained levels of arsenic, copper and tin two orders of magnitude higher than in unpolluted estuaries, while levels of silver and zinc were approximately forty times higher (Bryan & Gibbs, 1983). The presence of *Aphelochaeta marioni* in this area (Bryan & Gibbs, 1983) suggested that the species was tolerant of heavy metal contamination. Furthermore, *Aphelochaeta marioni* was shown to accumulate arsenic (Gibbs *et al.*, 1983). *Aphelochaeta marioni* (studied as *Tharyx marioni*) was found to have whole body concentrations of arsenic greater than 2000 μg/g dry weight (even when living under low ambient arsenic conditions). For reference, other Cirratulids, e.g. *Cirriformia tentaculata*, from the same habitat contained arsenic at concentrations lower than 100 μg/g dry weight. The purpose of arsenic accumulation was unclear. Trials with gobies failed to confirm that it was a predator deterrent mechanism and it is probably not a detoxification mechanism as arsenic accumulations were similar for worms living in widely varying arsenic concentrations. Hence, there is no evidence to suggest that *Aphelochaeta marioni* is intolerant of heavy metal contamination. However, other annelids have been shown to be intolerant of heavy metal contamination (e.g. see review by Crompton, 1997) and therefore an intolerance of low is recorded. Due to their tolerance, a recoverability of very high is recorded.

Sensitivity assessment. The review by Bryan (1984) suggested that polychaetes were "fairly resistant" to heavy metal pollution. The information on accumulation suggests that the species mentioned were resistant, while the evidence on copper in *Lumbrineris* suggests sensitivity. Overall, polychaetes are probably resistant but the level of resistance (or toxicity) varies between species, habitats and metals. Therefore, resistance is assessed as **'Medium'** as a precaution to represent a possible reduction in the abundance of some species within the community. Hence, resilience is assessed as **'High'** and sensitivity as **'Low'** but with 'Low' confidence due to the lack of direct evidence on the effects on the important characteristic species within this biotope.

Hydrocarbon & PAH contamination	Low	High	Low
	Q: Low A: NR C: NR	Q: High A: Medium C: Medium	Q: Low A: Low C: Low

Suchanek (1993) suggested that cirratulids were mostly immune to oil spills because their feeding tentacles are protected by a heavy secretion of mucus. This immunity is supported by observations of *Aphelochaeta marioni* following the Amoco Cadiz oil spill in March 1978 (Dauvin, 1982, 2000). Prior to the spill, *Aphelochaeta marioni* (studied as *Tharyx marioni*) was present in very low numbers in the Bay of Morlaix, western English Channel. Following the spill, the level of hydrocarbons in the sediment increased from 10

mg/kg dry sediment to 1443 mg/kg dry sediment six months afterwards. In the same period, *Aphelochaeta marioni* increased in abundance to a mean of 76 individuals per m, which placed it among the top five dominant species in the faunal assemblage. It was suggested that the population explosion occurred due to the increased food availability because of the accumulation of organic matter resulting from the high mortality of browsers. Six years later, an abundance of *Aphelochaeta marioni* began to fall away again, accompanied by gradual decontamination of the sediments. *Mediomastus fragilis* also increased in abundance (Dauvin, 2000).

George (1971) reported that the spawning, growth and mortality of *Cirriformia tentaculata* and *Cirratulus cirratus* were unaffected by a fuel oil spill on mudflats at the mouth of the River Hamble, England. However, the dispersants (Essolvene and BP1002) killed both species at low concentrations, although *Cirratulus cirratus* was the most tolerant. George (1971) reported 24-hour LC50 of 30 ppm BP1002 and 63 ppm Essolvene in *Cirriformia tentaculata* and 129 ppm and 162 ppm respectively in *Cirratulus cirratus*. Populations showed recovery two years after the pollution event.

Conan (1982) investigated the long-term effects of the *Amoco Cadiz* oil spill at St Efflam beach in France. *Fabulina fabula* (studied as *Tellina fabula*) started to disappear from the intertidal zone a few months after the spill and from then on was restricted to subtidal levels. In the following 2 years, recruitment of *Fabulina fabula* was very much reduced. The author commented that, in the long-term, the biotas most severely affected by oil spills are low-energy sandy and muddy shores, bays and estuaries. In such places, populations of species with long and short-term life expectancies (e.g. *Fabulina fabula*, *Echinocardium cordatum* and *Ampelisca* sp.) either vanished or displayed long-term decline following the *Amoco Cadiz* oil spill. However, polychaetes including *Nephtys hombergii*, cirratulids and capitellids were largely unaffected. Other studies support the conclusion that polychaetes are generally tolerant taxa. Hiscock *et al.* (2004; 2005a, from Levell *et al.*, 1989) described *Glycera* spp. as a very tolerant taxon, found in enhanced abundances in the transitional zone along hydrocarbon contamination gradients surrounding oil platforms.

The amphipods, *Ampelisca* sp. are also very intolerant of oil contamination and the recovery of the *Ampelisca* populations in the fine sand community in the Bay of Morlaix took up to 15 years following the *Amoco Cadiz* oil spill (Poggiale & Dauvin, 2001).

Sensitivity assessment. Overall, the characteristic polychaetes are probably resistant to exposure to oil spills and some species may increase in abundance due to the indirect effects of oil and resistance is assessed as **'High'**. Hence, resilience is **'High'** and sensitivity is assessed as **'Not sensitive' to oil spills.** However, dispersants have the potential to be more toxic. Therefore, resistance to dispersants is assessed as **'Low'** but with 'Low' confidence since it is based on a single study. Hence, resilience is 'High' and **sensitivity to dispersants** is assessed as **'Low'.**

Synthetic compound contamination	Medium	High	Low
	Q: Low A: NR C: NR	Q: High A: Medium C: Medium	Q: Low A: Low C: Low

The anti-parasite compound ivermectin is highly toxic to benthic polychaetes and crustaceans (Black *et al.*, 1997; Collier & Pinn, 1998; Grant & Briggs, 1998, cited in Wildling & Hughes, 2010). Data from a farm in Galway, Ireland indicated that ivermectin was detectable in sediments adjacent to the farm at concentrations up to 6.8 m/kg and to a depth of 9 cm (reported in OSPAR, 2000). Infaunal polychaetes have been affected by deposition rates of 78-780 mg ivermectin/m^2.

However, *Mediomastus californiensis* and other polychaetes (inc. *Nephtys* sp.) were little affected by the application of the pesticide carbaryl (at 5.6 kg/ha)) used to control mud shrimp on oyster beds in Willapa Bay, Washington, USA (Dumbauld *et al.*, 2001). However, the amphipods *Corophium acherusicum* and *Eohaustorius estuarius* suffered the highest short-term mortalities but recruited back within three weeks and were often more abundant on treated than in control sites a year after treatment (Dumbauld *et al.*, 2001).

Pridmore *et al.* (1992) reported that bivalves appeared to be the most affected by the application of the

organochlorine pesticide Chlordane to intertidal sandflats. The three most abundant bivalves *Chione stutchburyi*, *Tellina liliana* and *Nucula hartvigiana* declined in number by 31, 40 and 56% respectively. The burrowing capitellid *Heteromastus filiformis*, also declined in abundance by ca 48% while other polychaetes showed no significant change in abundance (Pridmore *et al.*, 1992).

Sensitivity assessment. The evidence is limited to the effects of a few chemicals on a few species that are congeners of species that occur in this biotope. The effects are likely to vary between species, habitat, and chemical as well as the concentration of chemical used and its dosing rate. There is evidence that ivermectin has affected polychaetes and non-target crustaceans adversely. However, polychaetes may be resistant to other pesticides while amphipods are not likely to be resistant to insecticides. Therefore, resistance is assessed as **'Medium'** to represent the potential for synthetic contaminants to affect some of the characteristic species. Hence, resilience is assessed as **'High'** and sensitivity as **'Low'** but with 'Low' confidence in the absence of more evidence.

Radionuclide contamination	No evidence (NEv)	No evidence (NEv)	No evidence (NEv)
	Q: NR A: NR C: NR	Q: NR A: NR C: NR	Q: NR A: NR C: NR

No evidence was found to support an assessment at the pressure benchmark. Following the Fukushima Dai-ichi nuclear power plant accident in August 2013, radioactive cesium concentrations in invertebrates collected from the seabed were assessed. Concentrations in bivalves and gastropods were lower than in polychaetes (Sohtome *et al.*, 2014). The data does not indicate that there were mortalities.

Introduction of other substances	No evidence (NEv)	Not relevant (NR)	No evidence (NEv)
	Q: NR A: NR C: NR	Q: NR A: NR C: NR	Q: NR A: NR C: NR

No evidence was found

De-oxygenation	Low	High	Low
	Q: Low A: NR C: NR	Q: High A: Medium C: Medium	Q: High A: Low C: Low

Riedel *et al.* (2012) assessed the response of benthic macrofauna to hypoxia advancing to anoxia in the Mediterranean. The hypoxic and anoxic conditions were created for 3-4 days in a box that enclosed in-situ sediments. In general, molluscs were more resistant than polychaetes, with 90% surviving hypoxia and anoxia, whereas only 10% of polychaetes survived. Exposed
individual *Timoclea ovata* and *Tellina serrata* survived the experiment but the exposed *Glycera* spp. died. In general, epifauna were more sensitive than infauna, mobile species more sensitive than sedentary species and predatory species more sensitive than suspension and deposit feeders. The test conditions did not lead to the production of hydrogen sulphide which may have reduced mortalities compared to some observations. In their review, Vaquer-Sunyer & Duarte (2008) concluded that crustaceans were more sensitive to hypoxia than polychaetes which were more sensitive than echinoderms while molluscs were amongst the most resistant.

Further evidence of sensitivity was available for some of the polychaete species associated with this biotope. Rabalais *et al.* (2001) observed that hypoxic conditions on the north Coast of the Gulf of Mexico (oxygen concentrations from 1.5 to 1 mg/l (1 to 0.7 ml/l) led to the emergence of *Lumbrineris* sp. from the substratum that then laid motionless on the surface. *Glycera alba* was found to be able to tolerate periods of anoxia resulting from inputs of organic-rich material from a wood pulp and paper mill in Loch Eil (Scotland) (Blackstock & Barnes, 1982). Nierman *et al.* (1990) reported changes in a fine sand community for the German Bight in an area with regular seasonal hypoxia. In 1983, oxygen levels were exceptionally low (<3 mg O$_2$/l) in large areas and <1 mg O$_2$/l in some areas. Species richness decreased by 30-50% and overall biomass fell. *Owenia fusiformis* were reduced in abundance significantly by the hypoxia *Spiophanes bombyx* was found in small numbers at some, but not all areas, during the period of hypoxia. Once oxygen

levels returned to normal *Spiophanes bombyx* increased in abundance; the evidence suggests that at least some individuals would survive hypoxic conditions.

Diaz & Rosenberg (1995) suggested that *Mediomastus ambiseta* and *Lumbrinereis verrilli* were resistant to moderate hypoxia. They reported that the effects of hypoxia in the Rappahannock River on communities dominated by opportunistic polychaetes were species-specific. For example, *Streblospio benedicti* and *Mediomastus ambiseta* became extinct locally after severe hypoxia. Similarly, a hypoxic event in the Gulf of Mexico in 1981 significantly reduced the abundance of otherwise dominant species such as *Mediomastus californiensis* and *Cirratulus filiformis*. Diaz & Rosenberg (1995) also suggested that *Ampelisca agassizi* and *Ampharete grubei* were sensitive to hypoxia.

Sensitivity assessment. Riedel *et al.* (2012) and Vaquer-Sunyer & Duarte (2008) provide evidence on general sensitivity trends. As the biotope is characterized by polychaetes, and in the absence of direct evidence, resistance is assessed as **'Low'** and resilience as **'High'** so that sensitivity is assessed as **'Low'**.

Nutrient enrichment	No evidence (NEv)	Not relevant (NR)	No evidence (NEv)
	Q: NR A: NR C: NR	Q: NR A: NR C: NR	Q: NR A: NR C: NR

This pressure relates to increased levels of nitrogen, phosphorus and silicon in the marine environment compared to background concentrations. The polychaetes and other associated invertebrate species are unlikely to be directly affected by changes in nutrient enrichment. The resultant additional growth of benthic microalgae may increase food for infaunal deposit-feeders. If an algal bloom was triggered by resultant eutrophication then it may result in organic enrichment of the sediment (see below) or a hypoxic event (see deoxygenation above). However, there is inadequate evidence to assess the direct effects of changes in nutrient levels at the benchmark level.

Organic enrichment	Medium	High	Low
	Q: Medium A: Medium C: Medium	Q: High A: Medium C: Medium	Q: Medium A: Medium C: Medium

Borja *et al.* (2000) and Gittenberger & Van Loon (2011) assigned *Aphelochaeta marioni* to AMBI Group IV: "second-order opportunistic species (slight to pronounced unbalanced situations); mainly small sized polychaetes: subsurface deposit feeders, such as cirratulids. *Mediomastus fragilis, Glycera alba, Glycera lapidum* and *Spiophanes bombyx* were characterized as AMBI Group III, defined as "species tolerant to excess organic matter enrichment; these species may occur under normal conditions, but their populations are stimulated by organic enrichment (slight unbalance situations)". *Lumbrineris latreilli* was characterized as AMBI Group II: "species indifferent to enrichment, always present in low densities with non-significant variations with time (from the initial state, to slight unbalance)". However, *Nucula nitidosa* was characterized as Group I: "species very sensitive to organic enrichment and present under unpolluted conditions (initial state)" (Borja *et al.*, 2000, Gittenberger & Van Loon, 2011).

Sensitivity assessment. At the pressure benchmark, organic inputs are likely to represent a food subsidy for the associated deposit-feeding species. Most of the characteristic polychaetes are unlikely to be significantly affected, while some species, e.g. *Nucula* sp. may be impacted. Therefore, resistance is assessed as **'Medium'**, resilience as **'High'** and sensitivity assessed as **'Low'**.

⚓ Physical Pressures

	Resistance	Resilience	Sensitivity
Physical loss (to land or freshwater habitat)	None	Very Low	High
	Q: High A: High C: High	Q: High A: High C: High	Q: High A: High C: High

All marine habitats and benthic species are considered to have a resistance of **'None'** to this pressure and to be unable to recover from a permanent loss of habitat (resilience is **'Very Low'**). Sensitivity within the direct spatial footprint of this pressure is, therefore **'High'**. Although no specific evidence is described, confidence in this assessment is 'High' due to the incontrovertible nature of this pressure.

Physical change (to another seabed type)	None	Very Low	High
	Q: High A: High C: High	Q: High A: High C: High	Q: High A: High C: High

The biotope is characterized by sediment (JNCC, 2022), so a change to an artificial or rock substratum would alter the character of the biotope leading to reclassification and the loss of the sedimentary community including the characterizing polychaetes and bivalves that live buried within the sediment.

Sensitivity assessment. Based on the loss of the biotope, resistance is assessed as **'None'**, resilience as **'Very Low'** (as the change at the pressure benchmark is permanent), and sensitivity is assessed as **'High'**.

Physical change (to another sediment type)	Low	Very Low	High
	Q: Low A: NR C: NR	Q: High A: High C: High	Q: Low A: Low C: Low

Sediment type is a key factor structuring the biological assemblage present in the biotope. Surveys over sediment gradients and before-and-after impact studies from aggregate extraction sites where sediments have been altered indicate patterns in change. The biotope classification (JNCC, 2015) provides information on the sediment types where biotopes are found and indicate likely patterns in change if the sediment were to alter.

Long-term alteration of sediment type to finer more unstable sediments was observed six years after aggregate dredging at moderate energy sites (Boyd *et al.*, 2005). The ongoing sediment instability was reflected in a biological assemblage composed largely of juveniles (Boyd *et al.*, 2005). Differences in biotope assemblages in areas of different sediment types are likely to be driven by pre and post-recruitment processes. Sediment selectivity by larvae will influence levels of settlement and distribution patterns. Snelgrove *et al.* (1999) demonstrated that *Spisula solidissima*, selected coarse sand over muddy sand, and capitellid polychaetes selected muddy sand over coarse sand, regardless of site. Both larvae selected sediments typical of adult habitats, however, some species were nonselective (Snelgrove *et al.*, 1999) and presumably in unfavourable habitats post recruitment, mortality will result for species that occur in a restricted range of habitats. Holme (1966) observed that *Glycymeris glycymeris* was absent from areas of the English Channel with finer sediments but was abundant in tidally-swept coarse areas. Some species may, however, be present in a range of sediments. Post-settlement migration and selectivity also occurred on small scales (Snelgrove *et al.*, 1999). Desprez (2000) found that a change of habitat to fine sands, from coarse sands and gravels (from deposition of screened sand following aggregate extraction), changed the biological communities present. *Tellina pygmaea* and *Nephtys cirrosa* dominated the fine sand community. Dominant species of coarse sands, *Echinocyamus pusillus* and *Amphipholis squamata*, were poorly represented and the characteristic species of gravels and shingles were absent (Desprez, 2000). Cooper *et al.* (2011) found that characterizing species from sand-dominated sediments were equally likely to be found in gravel-dominated sediments.

Sensitivity assessment. This biotope (SS.SMx.IMx.MedCirr) is found in mixed sediments, gravelly muddy sand, and muddy sandy gravel (JNCC, 2022). The change referred to in the pressure benchmark is a change in sediment classification (based on Long, 2006). For mixed sediments, resistance is assessed based on a change to either coarse (gravel-dominated) sediments or muds and sandy muds. A change in sediment type may not result in the loss of all the characterizing species but will affect the community composition and diversity and result in re-classification and, hence, loss of the biotope. A change to coarser sediment will probably result in a biotope similar to SS.SCS.CCS.MedLumVen, SS.SCS.ICS.MoeVen or SS.SCS.CCS.Pkef. A change to finer sediments may result in a biotope similar to SS.SMu.ISaMu.MysAbr or SS.SMu.ISaMu.MelMagThy. Therefore, resistance is assessed as **'Low'** as some species may remain.

Hence, biotope resilience is assessed as **'Very low'** (the pressure is a permanent change) and sensitivity as **'High'**.

Habitat structure changes - removal of substratum (extraction)	None	Medium	Medium
	Q: High A: High C: High	Q: High A: Medium C: Medium	Q: High A: Medium C: Medium

A number of studies assess the impacts of aggregate extraction on sand and gravel habitats. Recovery of sediments will be site-specific and will be influenced by currents, wave action and sediment availability (Desprez, 2000). Except in areas of mobile sands, the process tends to be slow (Kenny & Rees, 1996; Desprez, 2000). Boyd *et al.* (2005) found that in a site subject to long-term extraction (25 years), extraction scars were still visible after six years and sediment characteristics were still altered in comparison with reference areas with ongoing effects on the biota. The strongest currents are unable to transport gravel. A further implication of the formation of these depressions is a local drop in current strength associated with the increased water depth, resulting in the deposition of finer sediments than those of the surrounding substrata (Desprez *et al.*, 2000).

Sensitivity assessment. Resistance is assessed as **'None'** as the extraction of the sediment will remove the characterizing and associated species present, within the affected area. Resilience is assessed as **'Medium'** as some species may require longer than two years to re-establish (see resilience section) and sediments may need to recover (where exposed layers are different). Hence, sensitivity is assessed as **'Medium'.**

Abrasion/disturbance of the surface of the substratum or seabed	Medium	High	Low
	Q: Medium A: Medium C: Medium	Q: High A: Medium C: Medium	Q: Medium A: Medium C: Medium

Comparative studies between disturbed and undisturbed areas indicate that abrasion and disturbance from bottom trawling on coarse gravels, sands and muds reduce the abundance of organisms, biomass and species diversity (Collie *et al.*, 1997). For example, *Aphelochaeta marioni*, and *Lumbrineris latreilli* were characterized as AMBI Fisheries Review Group III "species insensitive to fisheries in which the bottom is disturbed; their populations do not show a significant decline or increase" (Gittenberger & Van Loon, 2011). *Mediomastus fragilis* was characterized as Group IV, "second-order opportunistic species, which are sensitive to fisheries in which the bottom is disturbed; their populations recover relatively quickly however and benefit from the disturbance, causing their population sizes to increase significantly in areas with intense fisheries". *Nucula nitidosa* was characterized as Group II, "species sensitive to fisheries in which the bottom is disturbed, but their populations recover relatively quickly" (Gittenberger & Van Loon, 2011). Bradshaw *et al.* (2000) noted that the abundance of *Lumbrinereis gracilis* and Cirratulidae indet. increased in experimental scallop dredged vs. undredged plots.

Sensitivity assessment. Abrasion is likely to damage epifauna and may damage a proportion of the characterizing species. Therefore, resistance is assessed as **'Medium'**. Hence, resilience is assessed as **'High'** as opportunistic species are likely to recruit rapidly and some damaged characterizing species may recover or recolonize and sensitivity is assessed as **'Low'.**

Penetration or disturbance of the substratum subsurface	Medium	High	Low
	Q: Medium A: Medium C: Medium	Q: High A: Medium C: Medium	Q: Medium A: Medium C: Medium

Comparative studies between disturbed and undisturbed areas indicate that abrasion and disturbance from bottom trawling on coarse gravels and sands, reduce the abundance of organisms, biomass and species diversity (Collie *et al.*, 1997). Undisturbed sites contain more calcareous tube worms, bryozoans and hydroids and small fragile polychaetes and brittlestars.

Experiments in shallow, wave-disturbed areas, using a toothed, clam dredge, found that deposit-feeding polychaetes were more impacted than carnivorous species. Dredging resulted in reductions of >90% of *Spiophanes bombyx* immediately post dredging compared with before impact samples and the population reduction persisting for 90 days (although results may be confounded by storm events within the monitoring period which caused sediment mobility). Some predatory polychaete taxa were enhanced by fishing. *Protodorvillea kefersteini* was one of these. Large increases in abundance in samples were detected post-dredging and persisting over 90 days. The passage of the dredge across the sediment floor will have killed or injured some organisms that will then be exposed to potential predators/scavengers (Frid *et al.*, 2000; Veale *et al.*, 2000) providing a food source to mobile scavengers including these species. *Protodorvillia kefersteini* also showed a rapid increase in abundance at 21 days after sediment disturbance (Thrush, 1986).

Bergman & Hup (1992) carried out a pre and post-experimental investigation using a 12 m beam trawl. The area was trawled three times over 2 days and samples were taken up to 2 weeks after trawling. Some benthic species showed a 10-65% reduction in density after trawling the area three times. There was a significant lowering of densities (40-60%) of echinoderms *Asterias rubens* and small *Echinocardium cordatum*, and of polychaete worms *Lanice conchilega* and *Spiophanes bombyx*. No change in the total density of *Owenia fusiformis* was observed (Bergman & Hup, 1992). Bradshaw *et al.* (2000) noted that the abundance of *Lumbrinereis gracilis* and Cirratulidae indet. increased in experimental scallop dredged vs. undredged plots.

Aphelochaeta marioni, and *Lumbrineris latreilli* were characterized as AMBI Fisheries Review Group III "species insensitive to fisheries in which the bottom is disturbed; their populations do not show a significant decline or increase" (Gittenberger & Van Loon, 2011). *Mediomastus fragilis* was characterized as Group IV, "second-order opportunistic species, which are sensitive to fisheries in which the bottom is disturbed; their populations recover relatively quickly however and benefit from the disturbance, causing their population sizes to increase significantly in areas with intense fisheries". *Nucula nitidosa* was characterized as Group II, "species sensitive to fisheries in which the bottom is disturbed, but their populations recover relatively quickly" (Gittenberger & Van Loon, 2011).

Sensitivity assessment. The trawling studies and Gittenberger & Van Loon (2011) suggest that the biological assemblage present in this biotope is characterized by species that are relatively resistant to penetration and disturbance of the sediments or recover quickly. Therefore, resistance is assessed as **'Medium'** as some species will be displaced and may be predated or injured and killed. Hence, resilience is assessed as **'High'** as most species will recover rapidly and sensitivity is assessed as **'Low'.**

Changes in suspended solids (water clarity)	High	High	Not sensitive
	Q: Low A: NR C: NR	Q: High A: High C: High	Q: Low A: Low C: Low

No direct evidence was found to assess impacts on the characterizing species. A change in turbidity at the pressure benchmark is assessed as an increase from intermediate 10-100 mg/l to medium (100-300 mg/l) and a change to clear (<10 mg/l). An increase or decrease in turbidity may affect primary production in the water column and indirectly alter the availability of phytoplankton food available to species in filter-feeding mode. However, phytoplankton will also be transported from distant areas and so the effect of increased turbidity may be mitigated to some extent.

Changes in turbidity and seston are not predicted to directly affect burrowing polychaetes that live within sediments. The biotope is dominated by deposit-feeding species that are unlikely to be directly affected by increases or decreases in suspended sediments. Therefore, biotope resistance is assessed as **'High'**, resilience as **'High'**, and sensitivity is assessed as **'Not sensitive'** albeit with 'Low' confidence.

Smothering and siltation rate changes (light)	High	High	Not sensitive
	Q: Low A: NR C: NR	Q: High A: High C: High	Q: Low A: Low C: Low

The addition of fine material will alter the character of this habitat by covering it with a layer of dissimilar

sediment and will reduce its suitability for the species associated with this feature. Recovery will depend on the rate of sediment mixing or removal of the overburden, either naturally or through human activities. Recovery to a recognisable form of the original biotope will not take place until this has happened. In areas where the local hydrodynamic conditions are unaffected, fine particles will be removed by wave action moderating the impact of this pressure. The rate of habitat restoration would be site-specific and would be influenced by the type of siltation and rate. The long-term or permanent addition of fine particles would lead to the reclassification of this biotope type (see physical change pressures).

Little direct evidence of the effects of smothering on the characteristic species was found. *Lumbrineris latreilli* was characterized as AMBI sedimentation Group III: "species insensitive to higher amounts of sedimentation, but don't easily recover from strong fluctuations in sedimentation" (Gittenberger & Van Loon, 2011). *Mediomastus fragilis*, *Aphelochaeta marioni* and *Nucula nitidosa* were characterized as Group IV, "second-order opportunistic species, insensitive to higher amounts of sedimentation; although they are sensitive to strong fluctuations in sedimentation, their populations recover relatively quickly and even benefit; this causes their population sizes to increase significantly in areas after a strong fluctuation in sedimentation" (Gittenberger & Van Loon, 2011). Ranchor (1976) suggested that *Nucula nitidosa* was tolerant of anaerobic conditions and their mobility, allowed them to survive when covered by sediments during stormy weather. Ranchor (1976) also reported that *Nucula nitidosa* was abundant in an area subject to sewage sludge dumping.

Sensitivity assessment. This biotope (SS.SMx.IMx.MedCirr) is exposed to strong to weak tidal streams in moderately wave exposed to sheltered conditions, so that fine sediments may be removed quickly. The biotope is dominated by infaunal deposit-feeding polychaetes and bivalves that are likely to survive short periods under 5 cm of rapidly deposited sediment. Therefore, resistance is assessed as **'High'**, resilience as **'High'** and sensitivity assessed as **'Not sensitive' at the benchmark level.**

Smothering and siltation rate changes (heavy)	Medium	High	Low
	Q: Low A: NR C: NR	Q: High A: Medium C: Medium	Q: Low A: Low C: Low

The addition of fine material will alter the character of this habitat by covering it with a layer of dissimilar sediment and will reduce its suitability for the species associated with this feature. Recovery will depend on the rate of sediment mixing or removal of the overburden, either naturally or through human activities. Recovery to a recognisable form of the original biotope will not take place until this has happened. In areas where the local hydrodynamic conditions are unaffected, fine particles will be removed by wave action moderating the impact of this pressure. The rate of habitat restoration would be site-specific and would be influenced by the type of siltation and rate. The long-term or permanent addition of fine particles would lead to the reclassification of this biotope type (see physical change pressures).

Little direct evidence of the effects of smothering on the characteristic species was found. *Lumbrineris latreilli* was characterized as AMBI sedimentation Group III: "species insensitive to higher amounts of sedimentation, but don't easily recover from strong fluctuations in sedimentation" (Gittenberger & Van Loon, 2011). *Mediomastus fragilis*, *Aphelochaeta marioni* and *Nucula nitidosa* were characterized as Group IV, "second-order opportunistic species, insensitive to higher amounts of sedimentation; although they are sensitive to strong fluctuations in sedimentation, their populations recover relatively quickly and even benefit; this causes their population sizes to increase significantly in areas after a strong fluctuation in sedimentation" (Gittenberger & Van Loon, 2011). Ranchor (1976) suggested that *Nucula nitidosa* was tolerant of anaerobic conditions and their mobility, allowed them to survive when covered by sediments during stormy weather. Ranchor (1976) also reported that *Nucula nitidosa* was abundant in an area subject to sewage sludge dumping.

Sensitivity assessment. This biotope (SS.SMx.IMx.MedCirr) is exposed to strong to weak tidal streams in moderately wave exposed to sheltered conditions, so that fine sediments may be removed quickly. The biotope is dominated by infaunal deposit-feeding polychaetes and bivalves that are likely to survive short periods of smothering by fine sediment. However, the deposition of 30 cm of fine sediment may take

several tidal cycles to be removed and some of the more sensitive species may be reduced in abundance within the affected area. Therefore, resistance is assessed as **'Medium'**, resilience as **'High'** and sensitivity assessed as **'Low' at the benchmark level.**

Litter	Not Assessed (NA)	Not assessed (NA)	Not assessed (NA)
	Q: NR A: NR C: NR	Q: NR A: NR C: NR	Q: NR A: NR C: NR

Not assessed.

Electromagnetic changes	No evidence (NEv)	No evidence (NEv)	No evidence (NEv)
	Q: NR A: NR C: NR	Q: NR A: NR C: NR	Q: NR A: NR C: NR

No evidence.

Underwater noise changes	Not relevant (NR)	Not relevant (NR)	Not relevant (NR)
	Q: NR A: NR C: NR	Q: NR A: NR C: NR	Q: NR A: NR C: NR

'Not relevant'.

Introduction of light or shading	Not relevant (NR)	Not relevant (NR)	Not relevant (NR)
	Q: NR A: NR C: NR	Q: NR A: NR C: NR	Q: NR A: NR C: NR

Invertebrate species such as the bivalves and polychaetes may possess rudimentary eyes and be able to perceive light and dark. Changes in light levels are unlikely to affect adult stages, especially burrowing, infaunal species. This pressure is therefore assessed as 'Not relevant'.

Barrier to species movement	Not relevant (NR)	Not relevant (NR)	Not relevant (NR)
	Q: NR A: NR C: NR	Q: NR A: NR C: NR	Q: NR A: NR C: NR

Not relevant. This pressure is considered applicable to mobile species, e.g. fish and marine mammals rather than seabed habitats. Physical and hydrographic barriers may limit the dispersal of larval stages or propagules. However, the dispersal of larval stages or propagules is not considered under the pressure definition and benchmark.

Death or injury by collision	Not relevant (NR)	Not relevant (NR)	Not relevant (NR)
	Q: NR A: NR C: NR	Q: NR A: NR C: NR	Q: NR A: NR C: NR

'Not relevant' to seabed habitats. NB. Collision by grounding vessels is addressed under 'surface abrasion'.

Visual disturbance	Not relevant (NR)	Not relevant (NR)	Not relevant (NR)
	Q: NR A: NR C: NR	Q: NR A: NR C: NR	Q: NR A: NR C: NR

'Not relevant'. Invertebrate species such as the bivalves and polychaetes may possess rudimentary eyes and be able to perceive light and dark but are unlikely to respond to visual disturbance as defined by this pressure.

🐛 Biological Pressures

	Resistance	Resilience	Sensitivity

Genetic modification & translocation of indigenous species

Not relevant (NR)	Not relevant (NR)	Not relevant (NR)
Q: NR A: NR C: NR	Q: NR A: NR C: NR	Q: NR A: NR C: NR

Key characterizing species within this biotope are not cultivated or translocated. This pressure is therefore considered 'Not relevant' to this biotope.

Introduction or spread of invasive non-indigenous species

Low	Very Low	High
Q: Low A: NR C: NR	Q: High A: High C: High	Q: Low A: NR C: NR

The American slipper limpet *Crepidula fornicata* was introduced to the UK and Europe in the 1870s from the Atlantic coasts of North America with imports of the eastern oyster *Crassostrea virginica*. It was recorded in Liverpool in 1870 and the Essex coast in 1887-1890. It has spread through expansion and introductions along the full extent of the English Channel and into the European mainland (Blanchard, 1997, 2009; Bohn *et al.*, 2012, 2013a, 2013b, 2015; De Montaudouin *et al.*, 2018; Helmer *et al.*, 2019; Hinz *et al.*, 2011; McNeill *et al.*, 2010; Powell-Jennings & Calloway, 2018; Preston *et al.*, 2020; Stiger-Pouvreau & Thouzeau, 2015).

Crepidula fornicata is recorded from shallow, sheltered bays, lagoons and estuaries or the sheltered sides of islands, in variable salinity (18 to 40) although it prefers ca 30 (Tillin *et al.*, 2020). Larvae require hard substrata for settlement. It prefers muddy gravelly, shell-rich, substrata that include gravel, or shells of other *Crepidula*, or other species e.g., oysters, and mussels. It is highly gregarious and seeks out adult shells for settlement, forming characteristic 'stacks' of adults. But it also recorded in a wide variety of habitats including clean sands, artificial substrata, *Sabellaria alveolata* reefs and areas subject to moderately strong tidal streams (Blanchard, 1997, 2009; Bohn *et al.*, 2012, 2013a, 2013b, 2015; De Montaudouin *et al.*, 2018; Hinz *et al.*, 2011; Powell-Jennings & Calloway, 2018; Preston *et al.*, 2020; Stiger-Pouvreau & Thouzeau, 2015; Tillin *et al.*, 2020).

High densities of *Crepidula fornicata* cause ecological impacts on sedimentary habitats. The species can form dense carpets that can smother the seabed in shallow bays, changing and modifying the habitat structure. At high densities, the species physically smothers the sediment, and the resultant build-up of silt, pseudofaeces, and faeces is deposited and trapped within the bed (Tillin *et al.*, 2020, Fitzgerald, 2007, Blanchard, 2009, Stiger-Pouvreau & Thouzeau, 2015). The biodeposition rates of *Crepidula* are extremely high and once deposited, form an anoxic mud, making the environment suitable for other species, including most infauna (Stiger-Pouvreau & Thouzeau, 2015, Blanchard, 2009). For example, in fine sands, the community is replaced by a reef of slipper limpets, that provide hard substrata for sessile suspension-feeders (e.g., sea squirts, tube worms and fixed shellfish), while mobile carnivorous microfauna occupy species between or within shells, resulting in a homogeneous *Crepidula* dominated habitat (Blanchard, 2009). Blanchard (2009) suggested the transition occurred and became irreversible at 50% cover of the limpet. De Montaudouin *et al.* (2018) suggested that homogenization occurred above a threshold of 20-50 *Crepidula* /m^2. Where slipper limpet stacks are abundant, few other bivalves can live amongst them (Fretter & Graham, 1981; Blanchard, 1997).

Impacts on the structure of benthic communities will depend on the type of habitat that *Crepidula* colonizes. De Montaudouin & Sauriau (1999) reported that in muddy sediment dominated by deposit-feeders, species richness, abundance and biomass increased in the presence of high densities of *Crepidula* (ca 562 to 4772 ind./m^2), in the Bay of Marennes-Oléron, presumably because the *Crepidula* bed provided hard substrata in an otherwise sedimentary habitat. In medium sands, *Crepidula* density was moderate (330-1300 ind./m^2) but there was no significant difference between communities in the presence of *Crepidula*. Intertidal coarse sediment was less suitable for *Crepidula* with only moderate or low abundances (11 ind./m^2) and its presence did not affect the abundance or diversity of macrofauna. However, there was a higher abundance of suspension–feeders and mobile Crustacea in the absence of *Crepidula* (De Montaudouin & Sauriau, 1999). The presence of *Crepidula* as an ecosystem engineer has created a range of new niche habitats, reducing biodiversity as it modifies habitats (Fitzgerald, 2007). De Montaudouin *et al.* (1999) concluded that

Crepidula did not influence macroinvertebrate diversity or density significantly under experimental conditions, on fine sands in Arcachon Bay, France. De Montaudouin *et al.* (2018) noted that the limpet reef increased the species diversity in the bed, but homogenised diversity compared to areas where the limpets were absent. In the Milford Haven Waterway (MHW), the highest densities of *Crepidula* were found in areas of sediment with hard substrata, e.g., mixed fine sediment with shell or gravel or both (grain sizes 16-256 mm) but, while *Crepidula* density increased as gravel cover increased in the subtidal, the reverse was found in the intertidal (Bohn *et al.*, 2015). Bohn *et al.* (2015) suggested that high densities of *Crepidula* in high-energy environments were possible in the subtidal but not the intertidal, suggesting the availability of this substratum type is beneficial for its establishment. Hinz *et al.* (2011) reported a substantial increase in the occurrence of *Crepidula* off the Isle of Wight, between 1958 and 2006, at a depth of ca 60 m, on hard substrata (gravel, cobbles, and boulders), swept by strong tidal streams. Presumably, *Crepidula* is more tolerant of tidal flow than the oscillatory flow caused by wave action which may be less suitable (Tillin *et al.*, 2020).

King scallop (*Pecten maximus*) and Queen scallop (*Aequipecten opercularis*) in the Bay of Brest, have been reported to decrease in the presence of *Crepidula*, largely due to silting and biodeposition that changes the habitat (Stiger Pouvreau & Thouzeau, 2015; Thouzeau *et al.*, 2000). The scallop post larvae are unable to settle and survive on muddy *Crepidula* substrata. *Crepidula* could potentially be the main competitor for *Pecten maximus*, specially creating competition for space (Menesguen & Gregoris, 2018; Ragueneau *et al.*, 2018). However, no direct competition for food was observed between *Crepidula* and the scallops (Thouzeau *et al.*, 2000, Chauvaud *et al.* 2000) and scallop shell growth rates did not decrease with increasing *Crepidula* populations. Therefore, although *Crepidula* populations will likely impact scallop post larvae settlement, it does not affect shell growth rates or adult survivorship (Thouzeau *et al.*, 2000). Models show that competition for space between the species does not impact the abundance of *Crepidula*, but does lower the abundance of Pecten sp. (Menesguen & Gregoris, 2018).

The colonial ascidian *Didemnum vexillum* is present in the UK but appears to be restricted to artificial surfaces such as pontoons, this species may, however, have the potential to colonize and smother offshore gravel habitats. Valentine *et al.* (2007) describe how *Didemnum* sp. appears to have rapidly colonized gravel areas on the Georges Bank (US/Canada boundary). Colonies can coalesce to form large mats that may cover more than 50% of the seabed in parts.

Sensitivity assessment. The sediments characterizing this biotope are likely to be too mobile and unsuitable for most of the invasive non-indigenous species currently recorded in the UK. The above evidence suggests that *Crepidula* could colonize mixed sediment habitats in the subtidal, typical of this biotope, due to the presence of gravel, shells, cobbles, or any other hard substrata that can be used for larvae settlement (Tillin *et al.*, 2020). Bohn *et al.* (2015) demonstrated that *Crepidula* had a preference for gravelly habitats, while De Montaudouin & Sauriau (1999) and Bohn *et al.* (2015) noted that *Crepidula* densities were low in intertidal coarse sediments. Therefore, *Crepidula* has the potential to colonize, and modify the habitat and its associated community due to the introduction of *Crepidula* shell biomass, silt, pseudofaeces and faeces (Blanchard, 2009; Tillin *et al.*, 2020), as occurs in maerl gravels (Grall & Hall-Spencer, 2003) resulting in the loss of the biotope. This is a moderately exposed to very sheltered habitat, so storms may mobilise the sediment (JNCC, 2022), which may also mitigate or prevent colonization by *Crepidula* at high densities, although it has been recorded from areas of strong tidal streams (Hinz *et al.*, 2011). Therefore, the habitat may be more suitable for *Crepidula* in wave sheltered areas of the biotope and where water movement is mediated by tidal flow rather than wave action, e.g., the deeper examples of the biotope. *Didemnum* sp. may also emerge as a threat to this biotope.

Therefore, resistance is assessed as **'Medium'** in examples where wave action is high and subject to storms but **'Low'** in wave sheltered areas dominated by tidal flow. Resilience is assessed as **'Very low'** as it would require the removal of *Crepidula*, probably by artificial means. Hence, sensitivity is assessed as **'High'** based on the worst-case scenario. *Crepidula* has not yet been reported to occur in this biotope so the confidence in the assessment is 'Low' and further evidence is required.

Introduction of microbial pathogens	No evidence (NEv)	Not relevant (NR)	No evidence (NEv)
	Q: NR A: NR C: NR	Q: NR A: NR C: NR	Q: NR A: NR C: NR

No evidence was found for the characterizing polychaete species.

Removal of target species	Not relevant (NR)	Not relevant (NR)	Not relevant (NR)
	Q: NR A: NR C: NR	Q: NR A: NR C: NR	Q: NR A: NR C: NR

The characteristic polychaete species are not directly targeted by fisheries. Hence, this pressure is assessed as **'Not relevant'.**

Removal of non-target species	Low	High	Low
	Q: Low A: NR C: NR	Q: High A: Medium C: Medium	Q: Low A: Low C: Low

Species within the biotope are not functionally dependent on each other, although biological interactions will play a role in structuring the biological assemblage through predation and competition. Removal of adults may support the recruitment of juveniles by reducing competition for space and consumption of larvae. Removal of species would also reduce the ecological services provided by these species such as secondary production and nutrient cycling.

Sensitivity assessment. Species within the biotope are relatively sedentary or slow-moving, although the infaunal position may protect some burrowing species from removal. Therefore, resistance is assessed as **'Low'** and resilience as **'High'**, as the habitat is likely to be directly affected by removal and some species will recolonize rapidly. Hence, sensitivity is assessed as **'Low'.**

Bibliography

Aberkali, H.B. & Trueman, E.R., 1985. Effects of environmental stress on marine bivalve molluscs. *Advances in Marine Biology*, **22**, 101-198.

Allen, P.L. & Moore, J.J. 1987. Invertebrate macrofauna as potential indicators of sandy beach instability. *Estuarine, Coastal and Shelf Science*, **24**, 109-125.

Ballarin, L., Pampanin, D.M. & Marin, M.G., 2003. Mechanical disturbance affects haemocyte functionality in the Venus clam *Chamelea gallina*. *Comparative Biochemistry and Physiology Part A: Molecular & Integrative Physiology*, **136** (3), 631-640.

Bergman, M.J.N. & Hup, M., 1992. Direct effects of beam trawling on macrofauna in a sandy sediment in the southern North Sea. *ICES Journal of Marine Science*, **49**, 5-11. DOI https://doi.org/10.1093/icesjms/49.1.5

Berrilli, F., Ceschia, G., De Liberato, C., Di Cave, D. & Orecchia, P., 2000. Parasitic infections of *Chamelea gallina* (Mollusca, Bivalvia) from commercially exploited banks of the Adriatic Sea. *Bulletin of European Association of Fish Pathologists*, **20** (5), 199-205.

Bijkerk, R., 1988. Ontsnappen of begraven blijven: de effecten op bodemdieren van een verhoogde sedimentatie als gevolg van baggerwerkzaamheden: literatuuronderzoek: RDD, Aquatic ecosystems.

Black, K.D., Fleming, S. Nickell, T.D. & Pereira, P.M.F. 1997. The effects of ivermectin, used to control sea lice on caged farmed salmonids, on infaunal polychaetes. *ICES Journal of Marine Science*, **54**, 276-279.

Blackstock, J. & Barnes, M., 1982. The Loch Eil project: biochemical composition of the polychaete, *Glycera alba* (Müller), from Loch Eil. *Journal of Experimental Marine Biology and Ecology*, **57** (1), 85-92.

Blanchard, M., 2009. Recent expansion of the slipper limpet population (*Crepidula fornicata*) in the Bay of Mont-Saint-Michel (Western Channel, France). *Aquatic Living Resources*, **22** (1), 11-19. DOI https://doi.org/10.1051/alr/2009004

Blanchard, M., 1997. Spread of the slipper limpet *Crepidula fornicata* (L.1758) in Europe. Current state and consequences. *Scientia Marina*, **61**, Supplement 9, 109-118. Available from: http://scimar.icm.csic.es/scimar/index.php/secId/6/IdArt/290/

Bohn, K., Richardson, C. & Jenkins, S., 2012. The invasive gastropod *Crepidula fornicata*: reproduction and recruitment in the intertidal at its northernmost range in Wales, UK, and implications for its secondary spread. *Marine Biology*, **159** (9), 2091-2103. DOI https://doi.org/10.1007/s00227-012-1997-3

Bohn, K., Richardson, C.A. & Jenkins, S.R., 2015. The distribution of the invasive non-native gastropod *Crepidula fornicata* in the Milford Haven Waterway, its northernmost population along the west coast of Britain. *Helgoland Marine Research*, **69** (4), 313.

Bohn, K., Richardson, C.A. & Jenkins, S.R., 2013a. Larval microhabitat associations of the non-native gastropod Crepidula fornicata and effects on recruitment success in the intertidal zone. *Journal of Experimental Marine Biology and Ecology*, **448**, 289-297. DOI https://doi.org/10.1016/j.jembe.2013.07.020

Bohn, K., Richardson, C.A. & Jenkins, S.R., 2013b. The importance of larval supply, larval habitat selection and post-settlement mortality in determining intertidal adult abundance of the invasive gastropod *Crepidula fornicata*. *Journal of Experimental Marine Biology and Ecology*, **440**, 132-140. DOI https://doi.org/10.1016/j.jembe.2012.12.008

Borja, A., Franco, J. & Perez, V., 2000. A marine biotic index to establish the ecological quality of soft-bottom benthos within European estuarine and coastal environments. *Marine Pollution Bulletin*, **40** (12), 1100-1114.

Boyd, S., Limpenny, D., Rees, H. & Cooper, K., 2005. The effects of marine sand and gravel extraction on the macrobenthos at a commercial dredging site (results 6 years post-dredging). *ICES Journal of Marine Science: Journal du Conseil*, **62** (2), 145-162.

Bradshaw, C., Veale, L.O., Hill, A.S. & Brand, A.R., 2000. The effects of scallop dredging on gravelly seabed communities. In: *Effects of fishing on non-target species and habitats* (ed. M.J. Kaiser & de S.J. Groot), pp. 83-104. Oxford: Blackwell Science.

Bryan, G.W. & Gibbs, P.E., 1983. *Heavy metals from the Fal estuary, Cornwall: a study of long-term contamination by mining waste and its effects on estuarine organisms.* Plymouth: Marine Biological Association of the United Kingdom. [Occasional Publication, no. 2.]

Bryan, G.W., 1984. Pollution due to heavy metals and their compounds. In *Marine Ecology: A Comprehensive, Integrated Treatise on Life in the Oceans and Coastal Waters*, vol. 5. *Ocean Management*, part 3, (ed. O. Kinne), pp.1289-1431. New York: John Wiley & Sons.

Cabioch, L., Dauvin, J.C. & Gentil, F., 1978. Preliminary observations on pollution of the sea bed and disturbance of sub-littoral communities in northern Brittany by oil from the *Amoco Cadiz*. *Marine Pollution Bulletin*, **9**, 303-307.

Capasso, E., Jenkins, S., Frost, M. & Hinz, H., 2010. Investigation of benthic community change over a century-wide scale in the western English Channel. *Journal of the Marine Biological Association of the United Kingdom*, **90** (06), 1161-1172.

Chauvaud, L., Jean, F., Ragueneau, O. & Thouzeau, G., 2000. Long-term variation of the Bay of Brest ecosystem: benthic-pelagic coupling revisited. *Marine Ecology Progress Series*, **200**, 35-48. DOI https://doi.org/10.3354/meps200035

Chícharo, L., Chícharo, M., Gaspar, M., Regala, J. & Alves, F., 2002. Reburial time and indirect mortality of *Spisula solida* clams caused by dredging. *Fisheries Research*, **59**, 247-257.

Collie, J.S., Escanero, G.A. & Valentine, P.C., 1997. Effects of bottom fishing on the benthic megafauna of Georges Bank. *Marine Ecology Progress Series*, **155**, 159-172. DOI https://doi.org/10.3354/meps155159

Collier, L.M. & Pinn, E.H., 1998. An assessment of the acute impact of the sea lice treatment Ivermectin on a benthic community. *Journal of Experimental Marine Biology and Ecology*, **230** (1), 131-147. DOI https://doi.org/10.1016/s0022-0981(98)00081-1

Conan, G., 1982. The long-term effects of the *Amoco Cadiz* oil spill. *Philosophical Transactions of the Royal Society of London B*, **297**, 323-333.

Cooper, K., Ware, S., Vanstaen, K. & Barry, J., 2011. Gravel seeding - A suitable technique for restoring the seabed following marine

aggregate dredging? *Estuarine, Coastal and Shelf Science*, **91** (1), 121-132.

Creutzberg, F., 1986. The distribution patterns of two bivalve species (*Nucula turgida, Tellina fabula*) along a frontal system in the southern North Sea. *Netherlands Journal of Sea Research*, **20**, 305-311.

Crompton, T.R., 1997. *Toxicants in the aqueous ecosystem*. New York: John Wiley & Sons.

Dauvin, J.C. & Gillet, P., 1991. Spatio-temporal variability in population structure of *Owenia fusiformis* Delle Chiaje (Annelida: Polychaeta) from the Bay of Seine (eastern English Channel). *Journal of Experimental Marine Biology and Ecology*, **152**, 105-122.

Dauvin, J.C., 1982. Impact of *Amoco Cadiz* oil spill on the muddy fine sand *Abra alba - Melinna palmata* community from the Bay of Morlaix. *Estuarine and Coastal Shelf Science*, **14**, 517-531.

Dauvin, J.C., 1985. Dynamics and production of a population of *Venus ovata* (Pennant) (Mollusca-Bivalvia) of Morlaix Bay (western English Channel). *Journal of Experimental Marine Biology and Ecology*, **91**, 109-123.

Dauvin, J.C., 1988a. Structure and trophic organization of the *Amphioxus lanceolatus - Venus fasciata* community from the Bay of Morlaix (Brittany). *Cahiers de Biologie Marine. Paris*, **29**, 163-185.

Dauvin, J.C., 2000. The muddy fine sand *Abra alba - Melinna palmata* community of the Bay of Morlaix twenty years after the *Amoco Cadiz* oil spill. *Marine Pollution Bulletin*, **40**, 528-536.

Davenport, J. & Davenport, J.L., 2005. Effects of shore height, wave exposure and geographical distance on thermal niche width of intertidal fauna. *Marine Ecology Progress Series*, **292**, 41-50.

Davis, J.P. & Wilson, J.G., 1983b. The population structure and ecology of *Nucula turgida* (Leckenby & Marshall) in Dublin Bay. *Progress in Underwater Science*, **8**, 53-60.

De Montaudouin, X. & Sauriau, P.G., 1999. The proliferating Gastropoda *Crepidula fornicata* may stimulate macrozoobenthic diversity. *Journal of the Marine Biological Association of the United Kingdom*, **79**, 1069-1077. DOI https://doi.org/10.1017/S0025315499001319

De Montaudouin, X., Andemard, C. & Labourg, P-J., 1999. Does the slipper limpet (*Crepidula fornicata* L.) impair oyster growth and zoobenthos diversity ? A revisited hypothesis. *Journal of Experimental Marine Biology and Ecology*, **235**, 105-124.

De Montaudouin, X., Blanchet, H. & Hippert, B., 2018. Relationship between the invasive slipper limpet *Crepidula fornicata* and benthic megafauna structure and diversity, in Arcachon Bay. *Journal of the Marine Biological Association of the United Kingdom*, **98** (8), 2017-2028. DOI https://doi.org/10.1017/s0025315417001655

Desprez, M., 2000. Physical and biological impact of marine aggregate extraction along the French coast of the Eastern English Channel: short- and long-term post-dredging restoration. *ICES Journal of Marine Science*, **57** (5), 1428-1438.

Desprez, M., Pearce, B. & Le Bot, S., 2010. The biological impact of overflowing sands around a marine aggregate extraction site: Dieppe (eastern English Channel). *ICES Journal of Marine Science*, **67**, 270-277. DOI https://doi.org/10.1093/icesjms/fsp245

Diaz, R.J. & Rosenberg, R., 1995. Marine benthic hypoxia: a review of its ecological effects and the behavioural responses of benthic macrofauna. *Oceanography and Marine Biology: an Annual Review*, **33**, 245-303.

Diaz-Castaneda, V., Richard, A. & Frontier, S., 1989. Preliminary results on colonization, recovery and succession in a polluted areas of the southern North Sea (Dunkerque's Harbour, France). *Scientia Marina*, **53**, 705-716.

Dittmann, S., 1999. Biotic interactions in a *Lanice conchilega* dominated tidal flat. In *The Wadden Sea ecosystem*, (ed. S. Dittmann), pp.153-162. Germany: Springer-Verlag.

Dumbauld, B.R., Brooks, K.M. & Posey, M.H., 2001. Response of an estuarine benthic community to application of the pesticide Carbaryl and cultivation of Pacific oysters (*Crassostrea gigas*) in Willapa Bay, Washington. *Marine Pollution Bulletin*, **42** (10), 826-844. DOI https://doi.org/10.1016/S0025-326X(00)00230-7

Emson, R.H., Jones, M. & Whitfield, P., 1989. Habitat and latitude differences in reproductive pattern and life-history in the cosmopolitan brittle-star *Amphipholis squamata* (Echinodermata). In: *Ryland, J.S., Tyler, P.A. (Eds.), Reproduction, Genetics and Distributions of Marine Organisms*, pp. 75-81. Olsen & Olsen, Fredensborg.

Essink, K., 1999. Ecological effects of dumping of dredged sediments; options for management. *Journal of Coastal Conservation*, **5**, 69-80.

Fahy, E., Carroll, J. & O'Toole, M., 2003. A preliminary account of fisheries for the surf clam *Spisula solida* (L) (Mactracea) in Ireland [On-line] http://www.marine.ie, 2004-03-16

Farke, H., 1979. Population dynamics, reproduction and early development of *Tharyx marioni* (Polychaeta, Cirratulidae) on tidal flats of the German Bight. *Veroffentlichungen des Instituts fur Meeresforschung in Bremerhaven*, **18**, 69-99.

FitzGerald, A., 2007. Slipper Limpet Utilisation and Management. Final Report. Port of Truro Oyster Management Group., Truro, 101 pp. Available from https://www.shellfish.org.uk/files/Literature/Projects-Reports/0701-Slipper_Limpet_Report_Final_Small.pdf

Folk, R.L., 1954. The distinction between grain size and mineral composition in sedimentary-rock nomenclature. **62**, *The Journal of Geology*, 344-359.

Ford, E., 1923. Animal communities of the level sea-bottom in the water adjacent to Plymouth. *Journal of the Marine Biological Association of the United Kingdom*, **13**, 164-224.

Fretter, V. & Graham, A., 1981. The Prosobranch Molluscs of Britain and Denmark. Part 6. Molluscs of Britain and Denmark. Part 6. *Journal of Molluscan Studies*, Supplement **9**, 309-313.

Frid, C.L., Harwood, K.G., Hall, S.J. & Hall, J.A., 2000. Long-term changes in the benthic communities on North Sea fishing grounds. *ICES Journal of Marine Science*, **57** (5), 1303.

Gaspar, M.B. & Monteiro, C.C., 1999. Gametogenesis and spawning in the subtidal white clam *Spisula solida*, in relation to temperature.

Journal of the Marine Biological Association of the United Kingdom, **79**, 753-755.

Gaspar, M.B., Leitão, F., Santos, M.N., Sobral, M., Chícharo, L., Chícharo, A. & Monteiro, C., 2002. Influence of mesh size and tooth spacing on the proportion of damaged organisms in the catches of the portuguese clam dredge fishery. *ICES Journal of Marine Science*, **59**,1228-1236.

Gaspar, M.B., Pereira, A.M., Vasconcelos, P. & Monteiro, C.C., 2004. Age and growth of *Chamelea gallina* from the Algarve coast (southern Portugal): influence of seawater temperature and gametogenic cycle on growth rate. *Journal of Molluscan Studies*, **70** (4), 371-377.

Gentil, F., Dauvin, J.C. & Menard, F., 1990. Reproductive biology of the polychaete *Owenia fusiformis* Delle Chiaje in the Bay of Seine (eastern English Channel). *Journal of Experimental Marine Biology and Ecology*, **142**, 13-23.

George, J.D., 1964a. The life history of the cirratulid worm, *Cirriformia tentaculata*, on the intertidal mudflat. *Journal of the Marine Biological Association of the United Kingdom*, **44**, 47-65.

George, J.D., 1964b. On some environmental factors affecting the distribution of *Cirriformia tentaculata* (Polychaete) at Hamble. *Journal of the Marine Biological Association of the United Kingdom*, **44**, 383-388.

George, J.D., 1968. The effect of the 1962-63 winter on the distribution of the cirratulid polychaetes, *Cirratulus cirratus* (Müller) and *Cirriformia tentaculata* (Montagu) in the British Isles. *Journal of Animal Ecology*, **37**, 321-31.

George, J.D., 1971. The effects of pollution by oil and oil dispersants on the common intertidal polychaetes, *Cirriformia tentaculata* and *Cirratulus cirratus*. *Journal of Applied Ecology*, **8**, 411-420.

Gibbs, P.E., 1971. Reproductive cycles in four polychaete species belonging to the family Cirratulidae. *Journal of the Marine Biological Association of the United Kingdom*, **51**, 745-769.

Gibbs, P.E., Burt, G.R., Pascoe, P.L., Llewellyn, C.A. & Ryan K.P., 2000. Zinc, copper and chlorophyll-derivates in the polychaete *Owenia fusiformis*. *Journal of the Marine Biological Association of the United Kingdom*, **80**, 235-248.

Gibbs, P.E., Langston, W.J., Burt, G.R. & Pascoe, P.L., 1983. *Tharyx marioni* (Polychaeta) : a remarkable accumulator of arsenic. *Journal of the Marine Biological Association of the United Kingdom*, **63**, 313-325.

Gilkinson, K., Paulin, M., Hurley, S. & Schwinghamer, P., 1998. Impacts of trawl door scouring on infaunal bivalves: results of a physical trawl door model/dense sand interaction. *Journal of Experimental Marine Biology and Ecology*, **224** (2), 291-312.

Gilkinson, K.D., Gordon, D.C., MacIsaac, K.G., McKeown, D.L., Kenchington, E.L., Bourbonnais, C. & Vass, W.P., 2005. Immediate impacts and recovery trajectories of macrofaunal communities following hydraulic clam dredging on Banquereau, eastern Canada. *ICES Journal of Marine Science: Journal du Conseil*, **62** (5), 925-947.

Giribet, G. & Peñas, A., 1999. Revision of the genus *Goodallia* (Bivalvia: Astartidae) with the description of two new species. *Journal of Molluscan Studies*, **65** (2), 251-265. DOI https://doi.org/10.1093/mollus/65.2.251

Gittenberger, A. & Van Loon, W.M.G.M., 2011. Common Marine Macrozoobenthos Species in the Netherlands, their Characterisitics and Sensitivities to Environmental Pressures. GiMaRIS report no 2011.08. DOI: 10.13140/RG.2.1.3135.7521

Glémarec, M., 1973. The benthic communities of the European North Atlantic continental shelf. *Oceanography and Marine Biology: an Annual Review*, **11**, 263-289.

Grall J. & Hall-Spencer J.M. 2003. Problems facing maerl conservation in Brittany. *Aquatic Conservation: Marine and Freshwater Ecosystems*, **13**, S55-S64. DOI https://doi.org/10.1002/aqc.568

Grant, A. & Briggs, A.D., 1998. Toxicity of Ivermectin to estuarine and marine invertebrates. *Marine Pollution Bulletin*, **36** (7), 540-541. DOI https://doi.org/10.1016/S0025-326X(98)00012-5

Guillou, J. & Sauriau, F.G., 1985. Some observations on the biology and ecology of a *Venus striatula* population in the Bay of Douarnenez, Brittany. *Journal of the Marine Biological Association of the United Kingdom*, **65**, 889-900.

Harvey, R. & Gage, J.D., 1995. Reproduction and recruitment of *Nuculoma tenuis* (Bivalvia: Nuculoida) from Loch Etive, Scotland. *Journal of Molluscan Studies*, **61**(4), 409-419.

Hauton, C., Hall-Spencer, J.M. & Moore, P.G., 2003. An experimental study of the ecological impacts of hydraulic bivalve dredging on maerl. *ICES Journal of Marine Science*, **60**, 381-392.

Helmer, L., Farrell, P., Hendy, I., Harding, S., Robertson, M. & Preston, J., 2019. Active management is required to turn the tide for depleted *Ostrea edulis* stocks from the effects of overfishing, disease and invasive species. *Peerj*, **7** (2). DOI https://doi.org/10.7717/peerj.6431

Hinz, H., Capasso, E., Lilley, M., Frost, M. & Jenkins, S.R., 2011. Temporal differences across a bio-geographical boundary reveal slow response of sub-littoral benthos to climate change. *Marine Ecology Progress Series*, **423**, 69-82. DOI https://doi.org/10.3354/meps08963

Hiscock, K., Langmead, O. & Warwick, R., 2004. Identification of seabed indicator species from time-series and other studies to support implementation of the EU Habitats and Water Framework Directives. *Report to the Joint Nature Conservation Committee and the Environment Agency from the Marine Biological Association.* Marine Biological Association of the UK, Plymouth. JNCC Contract F90-01-705. 109 pp.

Hiscock, K., Langmead, O., Warwick, R. & Smith, A., 2005. Identification of seabed indicator species to support implementation of the EU Habitats and Water Framework Directives. *Report to the Joint Nature Conservation Committee and the Environment Agency* The Marine Biological Association, Plymouth, 77 pp.

Hjulström, F., 1939. Transportation of detritus by moving water: Part 1. Transportation. Recent Marine Sediments, a Symposium (ed. P.D. Trask), pp. 5-31. Dover Publications, Inc.

Holme, N.A., 1966. The bottom fauna of the English Channel. Part II. *Journal of the Marine Biological Association of the United Kingdom*, **46**,

401-493.

JNCC (Joint Nature Conservation Committee), 2022. The Marine Habitat Classification for Britain and Ireland Version 22.04. [Date accessed]. Available from: https://mhc.jncc.gov.uk/

Joaquim, S., Gaspar, M.B., Matias, D., Ben-Hamadou, R. & Arnold, W.S., 2008. Rebuilding viable spawner patches of the overfished *Spisula solida* (Mollusca: Bivalvia): a preliminary contribution to fishery sustainability. *ICES Journal of Marine Science: Journal du Conseil*, **65** (1), 60-64.

Jones, N.S., 1950. Marine bottom communities. *Biological Reviews*, **25**, 283-313.

Jones, N.S., 1951. The bottom fauna of the south of the Isle of Man. *Journal of Animal Ecology*, **20**, 132-144.

Kühne, S. & Rachor, E., 1996. The macrofauna of a stony sand area in the German Bight (North Sea). *Helgoländer Meeresuntersuchungen*, **50** (4), 433.

Kaiser, M.J., & Spencer, B.E., 1994a. A preliminary assessment of the immediate effects of beam trawling on a benthic community in the Irish Sea. In *Environmental impact of bottom gears on benthic fauna in relation to natural resources management and protection of the North Sea.* (ed. S.J. de Groot & H.J. Lindeboom). NIOZ-Rapport, **11**, 87-94.

Kaschl, A. & Carballeira, A., 1999. Behavioural responses of *Venerupis decussata* (Linnaeus, 1758) and *Venerupis pullastra* (Montagu, 1803) to copper spiked marine sediments. *Boletin. Instituto Espanol de Oceanografia*, **15**, 383-394.

Kenny, A.J. & Rees, H.L., 1996. The effects of marine gravel extraction on the macrobenthos: results 2 years post-dredging. *Marine Pollution Bulletin*, **32** (8-9), 615-622.

Kenny, A.J. & Rees, H.L., 1994. The effects of marine gravel extraction on the macrobenthos: early post dredging recolonisation. *Marine Pollution Bulletin*, **28**, 442-447.

Kinne, O. (ed.), 1984. *Marine Ecology: A Comprehensive, Integrated Treatise on Life in Oceans and Coastal Waters.*Vol. V. *Ocean Management Part 3: Pollution and Protection of the Seas - Radioactive Materials, Heavy Metals and Oil.* Chichester: John Wiley & Sons.

Kirby, R.R., Beaugrand, G. & Lindley, J.A., 2008. Climate-induced effects on the meroplankton and the benthic-pelagic ecology of the North Sea. *Limnology and Oceanography*, **53** (5), 1805.

Klawe, W.L. & Dickie, L.M., 1957. Biology of the bloodworm, *Glycera dibranchiata* Ehlers, and its relation to the bloodworm fishery of the Maritime Provinces. *Bulletin of Fisheries Research Board of Canada*, **115**, 1-37.

Kröncke, I., Dippner, J., Heyen, H. & Zeiss, B., 1998. Long-term changes in macrofaunal communities off Norderney (East Frisia, Germany) in relation to climate variability. *Marine Ecology Progress Series*, **167**, 25-36.

Kranz, P.M., 1974. The anastrophic burial of bivalves and its paleoecological significance. *The Journal of Geology*, **82** (2), 237-265.

Le Bot, S., Lafite, R., Fournier, M., Baltzer, A. & Desprez, M., 2010. Morphological and sedimentary impacts and recovery on a mixed sandy to pebbly seabed exposed to marine aggregate extraction (Eastern English Channel, France). *Estuarine, Coastal and Shelf Science*, **89**, 221-233.

Leitão, F., Gaspar, M.B., Santos, M.N. & Monteiro, C.C., 2009. A comparison of bycatch and discard mortality in three types of dredge used in the Portuguese Spisula solida (solid surf clam) fishery. *Aquatic Living Resources*, **22** (1), 1-10.

Levell, D., Rostron, D. & Dixon, I.M.T., 1989. Sediment macrobenthic communities from oil ports to offshore oilfields. In *Ecological Impacts of the Oil Industry*, Ed. B. Dicks. Chicester: John Wiley & Sons Ltd.

Long, D., 2006. BGS detailed explanation of seabed sediment modified Folk classification. Available from: http://www.emodnet-seabedhabitats.eu/PDF/GMHM3_Detailed_explanation_of_seabed_sediment_classification.pdf

Lopez-Flores I., De la Herran, R., Garrido-Ramos, M.A., Navas, J.I., Ruiz-Rejon, C. & Ruiz-Rejon, M., 2004. The molecular diagnosis of *Marteilia refringens* and differentiation between *Marteilia* strains infecting oysters and mussels based on the rDNA IGS sequence. *Parasitology*, **19** (4), 411-419.

Mackie, A.S.Y., James, J.W.C., Rees, E.I.S., Darbyshire, T., Philpott, S.L., Mortimer, K., Jenkins, G.O. & Morando, A., 2006. BIOMÔR 4. The Outer Bristol Channel Marine Habitat Study. Studies in marine biodiversity and systematics from the National Museum of Wales, Cardiff. BIOMÔR Reports 4: 1–249 and A1–A227, + DVD-ROM (2007).

Mackie, A.S.Y., Oliver, P.G. & Rees, E.I.S., 1995. Benthic biodiversity in the southern Irish Sea. *Studies in Marine Biodiversity and Systematics from the National Museum of Wales. BIOMOR Reports*, no. 1.

Martínez, B., Arenas, F., Rubal, M., Burgués, S., Esteban, R., García-Plazaola, I., Figueroa, F., Pereira, R., Saldaña, L. & Sousa-Pinto, I., 2012. Physical factors driving intertidal macroalgae distribution: physiological stress of a dominant fucoid at its southern limit. *Oecologia*, **170** (2), 341-353.

Maurer, D., Keck, R.T., Tinsman, J.C., Leatham, W.A., Wethe, C., Lord, C. & Church, T.M., 1986. Vertical migration and mortality of marine benthos in dredged material: a synthesis. *Internationale Revue der Gesamten Hydrobiologie*, **71**, 49-63. DOI https://doi.org/10.1002/iroh.19860710106

McCall, P.L., 1977. Community patterns and adaptive strategies of the infaunal benthos of Long Island Sound. *Journal of Marine Research*, **35**, 221-266.

McNeill, G., Nunn, J. & Minchin, D., 2010. The slipper limpet *Crepidula fornicata* Linnaeus, 1758 becomes established in Ireland. *Aquatic Invasions*, **5** (Suppl. 1), S21-S25. DOI https://doi.org/10.3391/ai.2010.5.S1.006

Menesguen, A. & Gregoris, T., 2018. Modelling benthic invasion by the colonial gastropod *Crepidula fornicata* and its competition with the bivalve *Pecten maximus*. 1. A new 0D model for population dynamics of colony-forming species. *Ecological Modelling*, **368**, 277-287. DOI

https://doi.org/10.1016/j.ecolmodel.2017.12.005

Marine Ecological Surveys Limited (MES), 2008. *Marine Macrofauna Genus Trait Handbook.* Marine Ecological Surveys Limited: Bath.

MES, 2010. *Marine Macrofauna Genus Trait Handbook*. Marine Ecological Surveys Limited. http://www.genustraithandbook.org.uk/

Morton, B., 2009. Aspects of the biology and functional morphology of *Timoclea ovata* (Bivalvia: Veneroidea: Venerinae) in the Azores, Portugal, and a comparison with *Chione elevata* (Chioninae). *Açoreana*, **6**, 105-119.

Moulaert, I. & Hostens, K., 2007. Post-extraction evolution of a macrobenthic community on the intensively extracted Kwintebank site in the Belgian part of the North Sea. *CM Documents-ICES*, (A:12).

NBN, 2015. National Biodiversity Network 2015(20/05/2015). https://data.nbn.org.uk/

Nichols, F.H. & Thompson, J.K., 1985. Persistence of an introduced mudflat community in South San Francisco Bay, California. *Marine Ecology Progress Series*, **24**, 83-97.

Niermann, U., Bauerfeind, E., Hickel, W. & Westernhagen, H.V., 1990. The recovery of benthos following the impact of low oxygen content in the German Bight. *Netherlands Journal of Sea Research*, **25** (1), 215-226. DOI https://doi.org/10.1016/0077-7579(90)90023-A

OBIS (Ocean Biodiversity Information System), 2024. Global map of species distribution using gridded data. Available from: Ocean Biogeographic Information System. www.iobis.org. Accessed: 2024-01-04

Olafsson, E.B. & Persson, L.E., 1986. The interaction between *Nereis diversicolor* (Muller) and *Corophium volutator* (Pallas) as a structuring force in a shallow brackish sediment. *Journal of Experimental Marine Biology and Ecology*, **103**, 103-117.

Olafsson, E.B., Peterson, C.H. & Ambrose, W.G. Jr., 1994. Does recruitment limitation structure populations and communities of macro-invertebrates in marine soft sediments: the relative significance of pre- and post-settlement processes. *Oceanography and Marine Biology: an Annual Review*, **32**, 65-109

OSPAR, 2000. OSPAR decision 2000/3 on the use of organic-phase drilling fluids (OPF) and the discharge of OPF-contaminated cuttings. Summary Record OSPAR 2000. *OSPAR 00/20/1-E, Annex 18. Copenhagen, 26–30 June.*

Pearson, T.H. & Rosenberg, R., 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. *Oceanography and Marine Biology: an Annual Review*, **16**, 229-311.

Pedersen, M.F., Borum, J. & Fotel, L. F., 2009. Phosphorus dynamics and limitation of fast and slow-growing temperate seaweeds in Oslofjord, Norway. *Marine Ecology Progress Series*, **399**, 103-115

Petersen, C.G.J., 1918. The sea bottom and its production of fish food. A survey of the work done in connection with valuation of the Denmark waters from 1883-1917. *Report of the Danish Biological Station*, **25**, 1-62.

Poggiale, J.C. & Dauvin, J.C., 2001. Long term dynamics of three benthic *Ampelisca* (Crustacea - Amphipoda) populations from the Bay of Morlaix (western English Channel) related to their disappearance after the *Amoco Cadiz* oil spill. *Marine Ecology Progress Series*, **214**, 201-209.

Powell-Jennings, C. & Callaway, R., 2018. The invasive, non-native slipper limpet *Crepidula fornicata* is poorly adapted to sediment burial. *Marine Pollution Bulletin*, **130**, 95-104. DOI https://doi.org/10.1016/j.marpolbul.2018.03.006

Powilleit, M., Graf, G., Kleine, J., Riethmuller, R., Stockmann, K., Wetzel, M.A. & Koop, J.H.E., 2009. Experiments on the survival of six brackish macro-invertebrates from the Baltic Sea after dredged spoil coverage and its implications for the field. *Journal of Marine Systems*, **75** (3-4), 441-451.

Preston, J., Fabra, M., Helmer, L., Johnson, E., Harris-Scott, E. & Hendy, I.W., 2020. Interactions of larval dynamics and substrate preference have ecological significance for benthic biodiversity and *Ostrea edulis* Linnaeus, 1758 in the presence of *Crepidula fornicata*. *Aquatic Conservation: Marine and Freshwater Ecosystems*, **30** (11), 2133-2149. DOI https://doi.org/10.1002/aqc.3446

Price, H., 1982. An analysis of factors determining seasonal variation in the byssal attachment strength of *Mytilus edulis*. *Journal of the Marine Biological Association of the United Kingdom*, **62** (01), 147-155

Pridmore, R.D., Thrush, S.F., Cummings, V.J. & Hewitt, J.E., 1992. Effect of the organochlorine pesticide technical chlordane on intertidal macrofauna. *Marine Pollution Bulletin*, **24** (2), 98-102. DOI https://doi.org/10.1016/0025-326X(92)90737-Q

Rabalais, N.N., Harper, D.E. & Turner, R.E., 2001. Responses of nekton and demersal and benthic fauna to decreasing oxygen concentrations. In: *Coastal Hypoxia Consequences for Living Resources and Ecosystems*, (Edited by: Rabalais, N. N. and Turner, R. E.), *Coastal and Estuarine Studies 58*, *American Geophysical Union*, pp. 115–128. Washington D.C.

Rachor, E., 1976. Structure, dynamics and productivity of a population of *Nucula nitidosa* (Bivalvia, Protobranchiata) in the German Bight. *Berichte der Deutschen Wissenschaftlichen Kommission fur Meeresforschung*, **24**, 296-331.

Ragueneau, O., Raimonet, M., Maze, C., Coston-Guarini, J., Chauvaud, L., Danto, A., Grall, J., Jean, F., Paulet, Y. M. & Thouzeau, G., 2018. The Impossible Sustainability of the Bay of Brest? Fifty Years of Ecosystem Changes, Interdisciplinary Knowledge Construction and Key Questions at the Science-Policy-Community Interface. *Frontiers in Marine Science*, **5**. DOI https://doi.org/10.3389/fmars.2018.00124

Rhoads, D.C. & Young, D.K., 1970. The influence of deposit-feeding organisms on sediment stability and community trophic structure. *Journal of Marine Research*, **28**, 150-178.

Riedel, B., Zuschin, M. & Stachowitsch, M., 2012. Tolerance of benthic macrofauna to hypoxia and anoxia in shallow coastal seas: a realistic scenario. *Marine Ecology Progress Series*, **458**, 39-52.

Riera, R., Tuya, F., Ramos, E., Rodríguez, M. & Monterroso, Ó., 2012. Variability of macrofaunal assemblages on the surroundings of a brine disposal. *Desalination*, **291**, 94-100.

Roberts, D.A., Johnston, E.L. & Knott, N.A., 2010b. Impacts of desalination plant discharges on the marine environment: A critical review

of published studies. *Water Research*, **44** (18), 5117-5128.

Roche, C., Lyons, D.O.,O'Connor, B. 2007. *Benthic surveys of sandbanks in the Irish Sea*. Irish Wildlife Manuals, No. 29. National Parks and Wildlife Service, Department of Environment, Heritage and Local Government, Dublin, Ireland.

Rygg, B., 1985. Effect of sediment copper on benthic fauna. *Marine Ecology Progress Series*, **25**, 83-89.

Salzwedel, H., Rachor, E. & Gerdes, D., 1985. Benthic macrofauna communities in the German Bight. *Verifflithungen des Institut fur Meeresforschung in Bremerhaven*, **20**, 199-267.

Sardá, R., Pinedo, S. & Martin, D., 1999. *Seasonal dynamics of macroinfaunal key species inhabiting shallow soft-bottoms in the Bay of Blanes (NW Mediterranean)*. Publications Elsevier: Paris.

Sardá, R., Pinedo, S., Gremare, A. & Taboada, S., 2000. Changes in the dynamics of shallow sandy-bottom assemblages due to sand extraction in the Catalan Western Mediterranean Sea. *ICES Journal of Marine Science*, **57** (5), 1446-1453.

Savina, M. & Pouvreau, S., 2004. A comparative ecophysiological study of two infaunal filter-feeding bivalves: *Paphia rhomboıdes* and *Glycymeris glycymeris*. *Aquaculture*, **239** (1), 289-306.

Serrano, L., Cardell, M., Lozoya, J. & Sardá, R., 2011. A polychaete-dominated community in the NW Mediterranean Sea, 20 years after cessation of sewage discharges. *Italian Journal of Zoology*, **78** (sup1), 333-346.

Simboura, N. & Zenetos, A., 2002. Benthic indicators to use in ecological quality classification of Mediterranean soft bottom marine ecosystems, including a new biotic index. *Mediterranean Marine Science*, **3** (2), 77-111.

Sinderman, C.J., 1990. *Principle diseases of marine fish and shellfish, 2nd edition, Volume 2. Diseases of marine shellfish*. Academic Press, 521 pp.

Snelgrove, P.V., Grassle, J.P., Grassle, J.F., Petrecca, R.F. & Ma, H., 1999. In situ habitat selection by settling larvae of marine soft-sediment invertebrates. *Limnology and Oceanography*, **44** (5), 1341-1347.

Sohtome, T., Wada, T., Mizuno, T., Nemoto, Y., Igarashi, S., Nishimune, A., Aono, T., Ito, Y., Kanda, J. & Ishimaru, T., 2014. Radiological impact of TEPCO's Fukushima Dai-ichi Nuclear Power Plant accident on invertebrates in the coastal benthic food web. *Journal of Environmental Radioactivity*, **138**, 106-115.

Somaschini, A., 1993. A Mediterranean fine-sand polychaete community and the effect of the tube-dwelling *Owenia fusiformis* Delle Chiaje on community structure. *Internationale Revue de Gesamten Hydrobiologie*, **78**, 219-233.

Stiger-Pouvreau, V. & Thouzeau, G., 2015. Marine Species Introduced on the French Channel-Atlantic Coasts: A Review of Main Biological Invasions and Impacts. *Open Journal of Ecology*, **5**, 227-257. DOI https://doi.org/10.4236/oje.2015.55019

Stirling, E.A., 1975. Some effects of pollutants on the behaviour of the bivalve *Tellina tenuis*. *Marine Pollution Bulletin*, **6**, 122-124.

Suchanek, T.H., 1993. Oil impacts on marine invertebrate populations and communities. *American Zoologist*, **33**, 510-523. DOI https://doi.org/10.1093/icb/33.6.510

Thomas, R., 1975. Functional morphology, ecology, and evolutionary conservatism in the Glycymerididae (Bivalvia). *Palaeontology*, **18** (2), 217-254.

Thorson, G., 1946. Reproduction and larval development of Danish marine bottom invertebrates, with special reference to the planktonic larvae in the Sound (Øresund). *Meddelelser fra Kommissionen for Danmarks Fiskeri- Og Havundersögelser*, *Serie: Plankton*, **4**, 1-523.

Thorson, G., 1957. Bottom communities (sublittoral or shallow shelf). *Memoirs of the Geological Society of America*, **67**, 461-534.

Thouzeau, Gérard, Chauvaud, Laurent, Grall, Jacques & Guérin, Laurent, 2000. Rôle des interactions biotiques sur le devenir du pré-recrutement et la croissance de *Pecten maximus* (L.) en rade de Brest. *Comptes Rendus de l#&39;Académie des Sciences - Series III - Sciences de la Vie*, **323** (9), 815-825. DOI https://doi.org/10.1016/S0764-4469(00)01232-4

Thouzeau, G., Chavaud, L., Grall, J. & Guerin, L., 2000. Do biotic interactions control pre-recruitment and growth of *Pecten maximus* (L.) in the Bay of Brest ? *Comptes rendus - acadamies des sciences*, Paris, **323**, 815-825.

Tillin, H.M., Hiddink, J.G., Jennings, S. & Kaiser, M.J., 2006. Chronic bottom trawling alters the functional composition of benthic invertebrate communities on a sea-basin scale. *Marine Ecology Progress Series*, **318**, 31-45.

Tillin, H.M., Kessel, C., Sewell, J., Wood, C.A. & Bishop, J.D.D., 2020. Assessing the impact of key Marine Invasive Non-Native Species on Welsh MPA habitat features, fisheries and aquaculture. NRW Evidence Report. Report No: 454. Natural Resources Wales, Bangor, 260 pp. Available from https://naturalresourceswales.gov.uk/media/696519/assessing-the-impact-of-key-marine-invasive-non-native-species-on-welsh-mpa-habitat-features-fisheries-and-aquaculture.pdf

UKTAG, 2014. UK Technical Advisory Group on the Water Framework Directive [online]. Available from: http://www.wfduk.org

Vader, W.J.M., 1964. A preliminary investigation in to the reactions of the infauna of the tidal flats to tidal fluctuations in water level. *Netherlands Journal of Sea Research*, **2**, 189-222.

Valentine, P.C., Carman, M.R., Blackwood, D.S. & Heffron, E.J., 2007. Ecological observations on the colonial ascidian *Didemnum* sp. in a New England tide pool habitat. *Journal of Experimental Marine Biology and Ecology*, **342** (1), 109-121.

Van Dalfsen, J.A., Essink, K., Toxvig Madsen, H., Birklund, J., Romero, J. & Manzanera, M., 2000. Differential response of macrozoobenthos to marine sand extraction in the North Sea and the Western Mediterranean. *ICES Journal of Marine Science*, **57** (5), 1439-1445.

Vaquer-Sunyer, R. & Duarte, C.M., 2008. Thresholds of hypoxia for marine biodiversity. *Proceedings of the National Academy of Sciences*, **105** (40), 15452-15457.DOI https://doi.org/10.1073/pnas.0803833105

Vaudrey, J.M.P., Kremer, J.N., Branco, B.F. & Short, F.T., 2010. Eelgrass recovery after nutrient enrichment reversal. *Aquatic Botany*, **93** (4),

237-243.

Veale, L.O., Hill, A.S., Hawkins, S.J. & Brand, A.R., 2000. Effects of long term physical disturbance by scallop fishing on subtidal epifaunal assemblages and habitats. *Marine Biology*, **137**, 325-337.

Warwick, R.M. & Davis, J.R., 1977. The distribution of sublittoral macrofauna communities in the Bristol Channel in relation to the substrate. *Estuarine and Coastal Marine Science*, **5**, 267-288.

Widdows, J., Bayne, B.L., Livingstone, D.R., Newell, R.I.E. & Donkin, P., 1979. Physiological and biochemical responses of bivalve molluscs to exposure to air. *Comparative Biochemistry and Physiology*, **62A**, 301-308.

Wilding T. & Hughes D., 2010. A review and assessment of the effects of marine fish farm discharges on Biodiversity Action Plan habitats. S*cottish Association for Marine Science, Scottish Aquaculture Research Forum (SARF)*.

Wilson, J.G., 1992. Age specific energetics of reproduction in *Nucula turgida* (Leckenby & Marshall) a bivalve with lecithotrophic larval development. *Invertebrate Reproduction and Development*, **22**, 275-280.

Woodin, S.A., 1978. Refuges, disturbance and community structure: a marine soft bottom example. *Ecology*, **59**, 274-284.

Zühlke, R., 2001. Polychaete tubes create ephemeral community patterns: *Lanice conchilega* (Pallas, 1766) associations studied over six years. *Journal of Sea Research*, **46**, 261-272.

Zühlke, R., Blome, D., van Bernem, K.H. & Dittmann, S., 1998. Effects of the tube-building polychaete *Lanice conchilega* (Pallas) on benthic macrofauna and nematodes in an intertidal sandflat. *Senckenbergiana Maritima*, **29**, 131-138.