Polychaetes and *Angulus tenuis* in littoral fine sand

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

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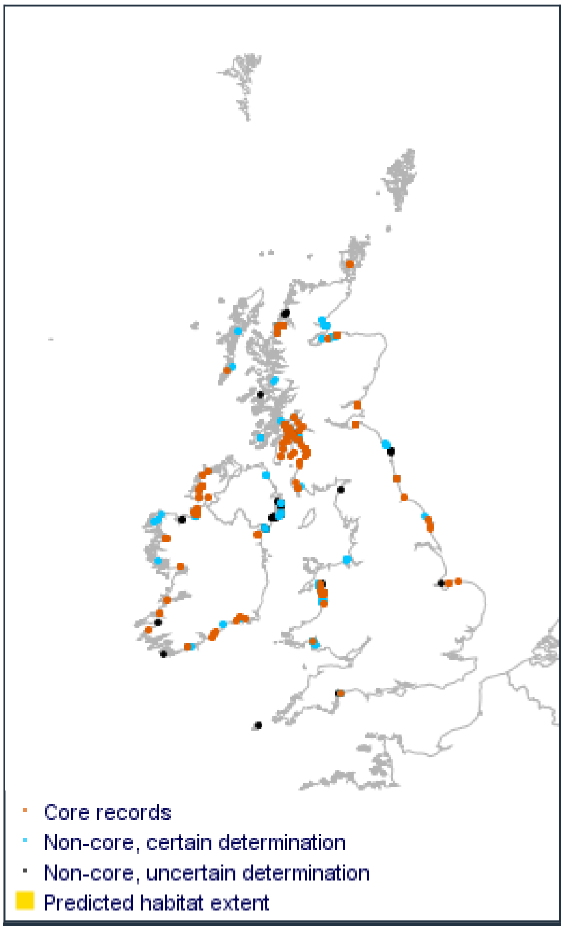

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Biotope distribution data provided by EMODnet Seabed Habitats (www.emodnet-seabedhabitats.eu)

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Summary

UK and Ireland classification

EUNIS 2008	A2.2312	Polychaetes and *Angulus tenuis* in littoral fine sand
JNCC 2015	LS.LSa.FiSa.Po.Aten	Polychaetes and *Angulus tenuis* in littoral fine sand
JNCC 2004	LS.LSa.FiSa.Po.Aten	Polychaetes and *Angulus tenuis* in littoral fine sand
1997 Biotope		

Description

This biotope occurs on the mid and lower shore on moderately wave-exposed and sheltered coasts, with predominantly fine sand which remains damp throughout the tidal cycle. The sediment is often rippled, and an anoxic layer may occasionally occur below a depth of 10 cm, though it is often patchy. The infaunal community is dominated by the abundant bivalve Angulus tenuis together with a range of polychaetes. The presence of polychaetes may be seen as coloured burrows running down from the surface of the sediment. Polychaetes that are characterizing for this biotope include *Nephtys cirrosa*, *Paraonis fulgens* and *Spio filicornis*. Burrowing amphipods

Bathyporeia spp. may occur in some samples of this biotope. (Information from Connor *et al*., 2004; JNCC, 2015).

Depth range

Mid shore, Lower shore

Additional information

-

Listed By

- none -

Further information sources

Search on:

G G JNCC

Sensitivity review

Sensitivity characteristics of the habitat and relevant characteristic species

The LS.LSa.FiSa.Po.Aten biotope occurs on the mid and lower shore on moderately wave-exposed and sheltered coasts. The infaunal community is dominated by the characterizing bivalve *Angulus* (now *Macoangulus*) *tenuis* together with a range of polychaetes. Polychaetes that characterize this biotope include *Nephtys cirrosa*, *Paraonis fulgens* and *Spio filicornis*. Burrowing amphipods *Bathyporeia* spp. may occur in some samples of this biotope.

The sensitivity assessments are based on the sensitivity of *Macoangulus tenuis*, the dominant polychaetes, and to a lesser extent the sensitivity of amphipods.

Resilience and recovery rates of habitat

The biotope can be subject to annual changes, as the infauna of the biotope may be reduced during winter, as increased storminess and wave action increases sediment mobility, leading to some species migrating or being washed out of the sediment. The biotope recovers during more stable periods through *in-situ* reproduction, although water transport of adults and active migration of mobile species such as *Nephtys cirrosa* and *Bathyporeia* spp. are likely to be the key recovery mechanisms, e.g. McLusky *et al*., 1983.

Little evidence was found to assess the recovery rates of the characterizing bivalve *Macoangulus tenuis*. In the Wadden Sea of northern Sylt, *Macoangulus tenuis* (recorded as *Angulus tenuis*) disappeared following a harsh winter in 1963 and by 1982, populations had not recovered, although no reason for this was suggested (Resise, 1982). The reduction in population would have reduced the available spat and it may be that the population had declined below limits that allowed successful annual recruitment. The population in the western Wadden Sea, however, increased during the 1970's following a succession of mild winters (Beukema *et al.* 1978).

Nephtys cirrosa 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-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).

Paraonis fulgens, is a small polychaete, up to 3 cm in length. *Paraonis fulgens* displays growth and reproduction strategies typical of opportunistic species and occurred in highly dynamic communities in German estuaries in a community of opportunistic species (Nehmer *et al*., 2003). Therefore, it is likely to show rapid recovery. *Paraonis fulgens* is thought to feed exclusively on benthic diatoms so that its abundance and recovery is likely to be affected by changes in levels of primary productivity (Gaston *et al*., 1992).

Spiophanes spp. (e.g. *Spiophanes filicornis*, *Spiophanes martinensis*, *Spiophanes bombyx*) have opportunistic life strategies (Kröencke, 1980; Niermann *et al*., 1990). They are characterized by small size, rapid maturation and short-lifespan of 1-2 years and produce large numbers of small propagules. It is often found at the early successional stages of variable, unstable habitats that it is quick to colonize following perturbation (Pearson & Rosenberg, 1978). For example, two years after dredging, the abundance of opportunistic species was generally elevated relative to pre-dredging levels and the communities were 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 from Boyd *et al.*, 2005) and their biomass was predicted to recover within 2-4 years.

Bathyporeia spp. are short lived, reaching sexual maturity within 6 months with 6-15 eggs per brood, depending on species. Reproduction may be continuous (Speybroeck *et al.*, 2008) with one set of embryos developing in the brood pouch whilst the next set of eggs is developing in the ovaries. However, specific reproductive periods vary between species and between locations (Mettam, 1989) and bivoltine patterns (twice yearly peaks in reproduction) have been observed (Mettam, 1989; Speybroeck *et al.*, 2008). Adult amphipods are highly mobile in the water column and recolonization by the adults is likely to be a significant recovery pathway. The life history traits of rapid sexual maturation and production of multiple broods annually support rapid local recolonization of disturbed sediments where some of the adult population remains.

Resilience assessment. This biotope is characterized by opportunistic polychaetes and mobile amphipods that are characteristic of biotopes subject to natural and/or anthropogenic disturbance. Biotope resilience is considered to be **High** as populations of the characterizing species are likely to recover within two years, even after severe depletion of the resident populations or community, unless the substratum or other key habitat factors are altered.

Hydrological Pressures

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

Intertidal species are exposed to extremes of high and low air temperatures during periods of emersion. They must also be able to cope with sharp temperature fluctuations over a short period of time during the tidal cycle. In winter air temperatures are colder than the sea, conversely in summer air temperatures are much warmer than the sea. Species that occur in the intertidal are therefore generally adapted to tolerate a range of temperatures, with the width of the thermal niche positively correlated with the height of the shore that the animal usually occurs at (Davenport & Davenport, 2005). The geographic distribution of species characteristic of this biotope extend south of the British Isles, further suggesting these species are likely to be resistant to an increase in temperature. Infaunal species are likely to be protected to some extent from direct effects of acute increases in temperature by sediment buffering, although increased temperatures may affect infauna indirectly by stimulating increased bacterial activity and increased oxygen consumption.

The characterizing bivalve *Macoangulus tenuis* is found off the Norwegian coasts to the Mediterranean and north-west coast of Africa and is likely to be resistant to temperature changes at the pressure benchmark.

Emery & Stevensen (1957) reported that *Nephtys spp.* could withstand summer temperatures of 30-35°C so is likely to withstand the benchmark acute temperature increase. An acute increase in temperature at the benchmark level may result in physiological stress endured by the infaunal species but is unlikely to lead to mortality. *Nephtys cirrosa* is an active worm that can swim short distances and, therefore, it could avoid short-term changes in temperature by migrating away from localised warmer spots.

No direct evidence was found to assess the sensitivity of *Paraonis fulgens*, however, this species is recorded in warmer waters than the UK in the Gulf of Mexico. *Paraonis fulgens* was one of the most abundant macrobenthic organisms collected in the shallow waters off Perdido Key, Florida, where winter water temperatures average 22 °C (Gaston *et al.* 1992). *Spiophanes bombyx* is found in the Mediterranean (Hayward & Ryland, 1995), which is likely to be warmer than the waters around Britain and Ireland.

The amphipods that occur within this habitat are mobile and can avoid unfavourable conditions to some extent. *Bathyporeia* life cycles vary between locations and this is related to temperature (Mettam, 1989). Preece (1971) tested temperature tolerances of *Bathyporeia pilosa* in the laboratory. Individuals acclimated to 15°C for 24 hours were exposed to temperature increases (water temperature raised by 0.2°C/minute). As test temperature were reached individuals were removed, placed in seawater at 4°C and allowed to recover for 24 hours at which point mortalities were tested. Amphipods were also allowed to bury into sediments and held at test temperatures for 24 hours of 32.5°C, 31.8°C and 29.5°C before being allowed to recover in fresh seawater at 15°C for a further 24 hours, before mortalities were assessed. Upper lethal temperatures (the temperature at which 50% of individuals died for adult males and gravid females of *Bathyporeia pilosa* were 39.4°C. These tests measures short-term exposure only and species had lower tolerance for longer-term (24 hour exposure). No mortality occurred for *Bathyporeia pilosa* individuals held at 29.5°C and 30.8°C; however 15% of individuals exposed to water temperatures of 31.8°C and 96% at 32.5°C died.

Sensitivity assessment. Typical surface water temperatures around the UK coast vary seasonally from 4-19°C (Huthnance, 2010). A chronic increase in temperature throughout the year of 2°C may fall within the normal temperature variation and an acute increase in water temperatures from 19 to 24°C for a month may be tolerated by the characterizing species supported by deeper burrowing and/or migration. It is likely that the characterizing species are able to resist a long-term increase in temperature of 2°C and may resist a short-term increase of 5°C. Resistance and resilience are, therefore assessed, as '**High'** and the biotope is assessed as '**Not Sensitive'** at the benchmark level

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

Intertidal species are exposed to extremes of high and low air temperatures during periods of emersion. They must also be able to cope with sharp temperature fluctuations over a short period of time during the tidal cycle. In winter air temperatures are colder than the sea, conversely in summer air temperatures are much warmer than the sea. Species that occur in the intertidal are therefore generally adapted to tolerate a range of temperatures, with the width of the thermal niche positively correlated with the height of the shore that the animal usually occurs at (Davenport & Davenport, 2005). Some of the characterizing species are found in colder waters that the UK suggesting these can tolerate colder waters than typically encountered. *Angulus tenuis* is found off the Norwegian coasts to the Mediterranean and north-west coast of Africa and is likely to be resistant to temperature changes at the pressure benchmark. In the Wadden Sea of northern Sylt, *Macoangulus tenuis* (recorded as *Angulus tenuis*) disappeared following a harsh winter in 1963 and by 1982, populations had not recovered, although no reason for this was suggested (Resise, 1982). The reduction in population would have reduced the available spat and it may be that the population had declined below limits that allowed successful annual recruitment. The population in the western Wadden Sea, however, increased during the 1970's following a succession of mild winters (Beukema *et al.* 1978). The temperatures during the severe winters exceeded -10°C for more than 5 days and thus are greater than the assessed pressure benchmark.

Paraonis fulgens occurs in colder waters than Irish and UK seas, such as the Bay of Fundy, Canada where winter temperatures are between 0 and 4 °C (Risk & Tunnicliffe 2006). *Spiophanes bombyx* is found in water off Denmark (Thorson, 1946) which are likely to be colder than British and Irish waters.However, *Nephtys cirrosa* reaches its northern limit in Scotland, and German Bight of the North Sea. A decrease in temperature may result in loss of the species from the biotope in these areas.

Crisp (1964) reported that species of amphipod seemed to be unharmed by the severe winter of 1962-1963. This may be due to burial in sediments buffering temperature or seasonal migration to deeper waters to avoid freezing. In the winter migrations have also been observed for *Bathyporeia* spp. (Fish & Fish, 1978; Fish & Preece, 1970). Preece (1971) tested temperature tolerances of *Bathyporeia pilosa* in the laboratory. Individuals acclimated to 15°C for 24 hours were placed in a freezer in wet sediment. As test temperatures were reached individuals were removed and allowed to recover for 24 hours at which point mortalities were tested. Amphipods were also allowed to bury into sediments and held at test temperatures of -1°C, -3°C and -5°C for 24 hours before being allowed to recover in fresh seawater at 15°C for a further 24 hours before mortalities were assessed. The lower lethal short-term tolerances of *Bathyporeia pilosa* were -13.6°C. *Bathyporeia pilosa* individuals could withstand temperatures as low as -1°C for 24 hours, at -3°C, 5% of *Bathyporeia pilosa* died but this rose to 82% at -5°C.

Sensitivity assessment..Typical surface water temperatures around the UK coast vary seasonally from 4-19°C (Huthnance, 2010). A chronic decrease in temperature throughout the year of 2°C may fall within the normal temperature variation but an acute decrease in water temperatures from 4°C to -1°C at the coldest part of the year may lead to freezing and lethal effects but may be tolerated by the characterizing species through deeper burrowing and/or migration. However, the abundance of *Nephtys cirrosa* may be reduced in northern examples of the biotope or severe winters. Therefore, biotope resistance is assessed as **Medium**. However, resilience is probably '**High'** and sensitivity is assessed as **Low**.

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

This biotope is found in full salinity (30-35 ppt) and variable salinity habitats (18-35 ppt) (JNCC, 2015). A change at the pressure benchmark is therefore assessed as a change to hypersaline conditions (>40 ppt) from full salinity. Little evidence was found to assess responses to hypersalinity and no evidence was found for *Macoangulus tenuis* which is typical of estuarine conditions.

Monitoring at a Spanish desalination facility where discharges close to the outfall reached a salinity of 53, found that amphipods were sensitive to the increased salinity and that species free-living in the sediment were most sensitive (De-la-Ossa-Carretero *et al*., 2016). Roberts *et al.* (2010) concluded that the reported effects of brine discharges were limited and difficult to compare but identified some trends. Hypersaline effluents tend to disperse quickly in well flushed environments like the habitat this biotope occurs in. However, sediment communities were affected in the immediate vicinity of brine discharges. For example, one of the studies reviewed found that the sediment became dominated by nematodes, with polychaetes, crustaceans and molluscs only fond at a distance from the outfall. Another study noted that the diversity of polychaete communities decreased adjacent to the outfall, and that the Ampharetidae were the most sensitive while the Paranoidae were the least sensitive.

Sensitivity assessment. No direct evidence was found to assess biotope sensitivity. However, if the biotope was exposed to hypersaline effluents then a proportion of the community may be lost and species diversity and abundances are likely to decrease. Therefore, a biotope resistance of **Low** is suggested. Resilience is probably **High** (following restoration of the usual salinity regime) so that sensitivity is assessed as **Low**.

Salinity decrease (local)	High Q: High A: Medium C: Medium	High Q: High A: Low C: Medium	Not sensitive Q: High A: Low C: Medium

The biotope occurs in full and variable salinity. Intertidal flats are likely to experience short periods of reduced salinities following rainfall at low tide. However, freshwater sits on the surface of denser seawater and interstitial water remains close to full salinity. Species will have some tolerances for salinity change, and may osmoregulate, may stop irrigating their burrow, or may move seaward if mobile or burrow deeper into the sediment (McLusky, 1989). Short-term tolerance is likely to be high but changes in salinity over the longer time may affect the biological assemblage. A decrease in salinity at the pressure benchmark refers to a change to low salinity (<18ppt) for a year and is likely to lead to a change in biotope character and classification as the reduction is likely to exceed some species tolerances.

Nephtys cirrosa were most abundant in salinities >30 psu in the German Bight (south eastern North Sea) (Meißner *et al.*, 2008), while *Spiophanes bombyx* is a euryhaline species (Bailey-Brook, 1976; Maurer & Lethem, 1980), inhabiting fully saline and estuarine habitats. *Spio martinensis* was a characterizing species in estuarine channels and inlets in Germany, suggesting resistance to lower salinities (Nehmer *et al.*, 2003)

Mobile species, such as *Nepthys cirrosa* and the amphipods may migrate to avoid unfavourable conditions. *Bathyporeia pelagica* migrates seaward in response to reduced salinities, the effect of which is enhanced by higher temperature (Preece, 1970). *Bathyporeia pilosa* is, however, more tolerant than *Bathyporeia pelagica* of low salinities and is capable of reproducing at salinities as low as 2 (Khayrallah, 1977). Populations of *Bathyporeia pilosa* within the upper reaches of the Severn Estuary experience wide fluctuations in salinity ranging from 1-22 depending on the season and tidal cycle (Mettam, 1989). The physiological stress for this environment affects size and reproduction (Mettam, 1989). Speybroeck *et al.* (2008) noted that *Bathyporeia pilosa* tends to occur subtidally in estuarine and brackish conditions. Local populations may be acclimated to the prevailing salinity regime and may exhibit different tolerances to other populations subject to different salinity conditions and, therefore, caution should be used when inferring tolerances from populations in different regions.

Sensitivity assessment. A decrease in salinity at the pressure benchmark is may lead to some changes in species presence and abundance and the loss of characterizing species such as *Macoangulus tenuis* and *Nephtys cirrosa*. *Bathyporeia pilosa* has high salinity tolerances and may replace more sensitive species, oligochaetes tolerant of low salinity conditions may also occur. based on the loss of the characteristic biotope, resistance is assessed as 'Low' over the course of a year and recovery is assessed as 'High' following re-establishment of typical habitat conditions. Biotope sensitivity is therefore assessed as 'Low'. Following the loss of populations of *Macoangulus* (*as Angulus*) *tenuis* in the Wadden Sea following a series of severe winters, recovery of *Macoangulus tenuis* was not observed over the corse of >5 years (Reise, 1982) and the possibility of prolonged recovery and higher sensitivity should be considered by managers if changes would affect a large area.

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

The biotope and sub biotopes occur on moderately exposed or sheltered beaches. Tidal flow velocities are likely to be variable across the range of habitats where this biotope occurs. Changes in flow velocity may lead to changes to another sub-biotope within the LS.LSa.FiSa.Po group.

Sensitivity assessment. The LS.LSa.FiSa.Po biotope (and sub-biotopes) occur in water flow velocities from very weak to moderately strong and in a moderately strong to sheltered wave climate The fine sand in low in mud and organic matter, which suggests it is well sorted and oxygenated by water movement. Species characterizing the biotope are likely to be resistant to changes at the pressure benchmark level (a 0.1-0.2 m/s change in flow,) although changes between sub-biotopes may occur if silt or mud content of the substratum changes. Resistance and resilience are assessed as '**High**' and sensitivity is assessed as '**Not Sensitive**'

Emergence regime changes	Medium Q: Low A: NR C: NR	High Q: High A: Low C: Medium	Low Q: Low A: Low C: Low

The biotope LS.LSa.FiSa.Po.Aten is typically found on the mid and lower shore. An increase in emergence may result in loss of the upper extent of the biotope and replacement by the similar biotope LS.LSa.FiSa.Po.Pfu, LS.LSa.MoSa.AmScoEur or LS.LSa.MoSa.Ol.FS which tend to occur higher on the shore (JNCC, 2015). A decrease in emergence may allow the biotope to extend up the shore if suitable habitat exists. However a decrease in emergence may result in decreased recruitment and survival of *Macoangulus tenuis* due to predation by shrimp, juvenile flatfish and other predators .

Sensitivity assessment. Increased and decreased emergence are likely to reduce the upper and lower limits of the biotope, respectively. Although polychaetes and amphipods would probably migrate down the shore, the upper or lower extent of the biotope may be replaced by a species assemblage more typical of the changed shore level. Therefore, a resistance of **Medium** is suggested (as 25-75% of the biotope may be lost). Resilience is probably **High** (following restablishment of emergence regime) and sensitivity is assessed as **Low**.

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

The biotope occur on shores that range from moderately exposed to extremely sheltered (JNCC, 2015). Increases and decreases in wave exposure may lead to increased erosion or deposition.

Increased wave exposure is likely to resuspend finer material and may lead to loss or reduction in abundance of *Macoangulus tenuis* and a greater abundance of *Nepthys cirrosa* which is characteristic of the coarser sand biotope LS.LSa.FiSa.Po.Ncir. The circulatory motion of wave action may also wash infauna such as *Nepthys cirrosa* and *Macoangulus tenuis* from the sediment in the most exposed locations. Although increased wave action is likely to wash some individuals from the sediment, recovery would be rapid.

Sensitivity assessment. An increase in wave height at the benchmark level is unlikely to create a noticeable impact, where initial conditions are sheltered. The biotope is reported to be naturally disturbed by winter storms (Connor *et al*., 2004) and a 3-5% change in significant wave height (the benchmark) is unlikely to affect the biotope adversely Therefore, resistance and resilience are assessed as '**High**', and the biotope is assessed as , '**Not Sensitive'** at the benchmark level.

Chemical Pressures

	Resistance	Resilience	Sensitivity
Transition elements & organo-metal contamination	Not Assessed (NA) Q: NR A: NR C: NR	Not assessed (NA) Q: NR A: NR C: NR	Not assessed (NA) Q: NR A: NR C: NR

This pressure is **Not assessed** but evidence is presented where available.

Levels of contaminants that exceed the pressure benchmark may cause impacts. Bryan & Gibbs (1983) reported lower sediment-metal concentrations in sandy areas than mud near the mouth of Restronguet Creek, a branch of the Fal Estuary system which is heavily contaminated with metals. Although heavy metals may not accumulate in the substratum to the extent that they would in muddy substrata, characterizing infauna are likely to be susceptible. Bryan & Gibbs (1983) suggested that in populations of polychaetes exposed to heavy metal contamination for a long period, metal resistance could be acquired. For example *Nephtys hombergii* from Restronguet Creek seemed able to regulate copper. The head end of the worm became blackened and x-ray microanalysis by Bryan & Gibbs (1983) indicated that this was caused by the deposition of copper sulphide in the body wall. In the same study, Bryan & Gibbs (1983) presented evidence that *Nephtys hombergii* from Restronguet Creek possessed increased tolerance to copper contamination. Specimens from the Tamar Estuary had a 96 h LC50 of 250 µg/l, whilst those from Restronguet Creek had a 96 h LC50 of 700 µg/l (35 psu; 13°C).

Hydrocarbon & PAH contamination	Not Assessed (NA) Q: NR A: NR C: NR	Not assessed (NA) Q: NR A: NR C: NR	Not assessed (NA) Q: NR A: NR C: NR

This pressure is **Not assessed** but evidence is presented where available.

Contamination at levels greater than the pressure benchmark may adversely influence the biotope. Suchanek (1993) reviewed the effects of oil spills on marine invertebrates and concluded that, in general, on soft sediment habitats, infaunal polychaetes, bivalves and amphipods were particularly affected.

Oil spills resulting from tanker accidents have caused deterioration of sandy communities in the intertidal and shallow sublittoral. Subtidal sediments, however, may be at less risk from oil spills unless oil dispersants are used, or if wave action causes dispersion of oil into the water column and sediment mobility drives oil in to the sediment (Elliott *et al*., 1998). Microbial degradation of the oil within the sediment would increase the biological oxygen demand and oxygen within the sediment may become significantly reduced.

Species within the biotope have been reported to be intolerant of oil pollution, e.g. amphipods (Suchanek, 1993). After the *Amoco Cadiz* oil spill there was a reduction in both the number of amphipod species and the number of individuals (Cabioch *et al.*, 1978). Initially, significant mortality would be expected, attributable to toxicity. Amphipod populations have been reported not return to pre-spill abundances for five or more years, which is most likely related to the persistence of oil within sediments (Southward, 1982). *Nephtys* species were amongst the fauna that was eradicated from sediments following the 1969 West Falmouth spill of Grade 2 diesel fuel documented by Sanders (1978).

Multivariate analysis showed that the *Prestige* oil spill scarcely affected the macroinfaunal community structure during the study period (2003-2009) and its effect was limited just to the first campaign (2003), six months after the *Prestige* accident (Junoy *et al.*, 2013). Opportunistic species such *Capitella capitata* have been shown to increase in abundance close to sources of contamination. High numbers of *Capitella capitata* have been recorded in hydrocarbon contaminated sediments (Ward & Young, 1982; Olsgard, 1999; Petrich & Reish, 1979) and colonization of areas defaunated by high hydrocarbon levels may be rapid (Le Moal, 1980). After a major spill of fuel oil in West Virginia *Capitella capitata* increased dramatically alongside large increases in *Polydora ligni* and *Prionospio* sp. (Sanders *et al.* 1972, cited in Gray 1979).

Synthetic compound contamination	Not Assessed (NA) Q: NR A: NR C: NR	Not assessed (NA) Q: NR A: NR C: NR	Not assessed (NA) Q: NR A: NR C: NR

This pressure is **Not assessed** but evidence is presented where available.

Boon *et al.* (1985) reported that *Nephtys* species in the North Sea accumulated organochlorines but, based on total sediment analyses, organochlorine concentrations in *Nephtys* species were not correlated with the concentrations in the (type of) sediment which they inhabited.

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

No evidence was found to assess this pressure.

Introduction of other substances	Not Assessed (NA) Q: NR A: NR C: NR	Not assessed (NA) Q: NR A: NR C: NR	Not assessed (NA) Q: NR A: NR C: NR

This pressure is **Not assessed**.

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

No information concerning the reduced oxygen tolerance of *Nephtys cirrosa* was found but evidence (Alheit, 1978; Arndt & Schiedek, 1997; Fallesen & Jørgensen, 1991) indicated a similar species, *Nephtys hombergii*, to be very tolerant of episodic oxygen deficiency and at the benchmark duration of one week. *Nephtys cirrosa* and *Spio spp.* were classified by Borja *et al.* (2000) as being indifferent to enrichment, suggesting some resilience to de-oxygenation. Dense *Capitella capitata* populations are frequently located in areas with greatly elevated organic content, even though eutrophic sediments are often anoxic and highly sulfidic (Tenore 1977; Warren 1977;

Tenore & Chesney 1985; Bridges et al. 1994). The polychaetes *Capitella capitata*, *Pygospio elegans* and *Scoloplos armiger* have all been reported to recolonize habitats following periods of anoxia and hypoxia.

Sensitivity assessment. The species characterizing the biotope are mobile and able to migrate vertically or horizontally to escape unsuitable conditions. The biotope is characterized by well sorted and oxygenated sands, where the anoxic layer occurs below 10 cm and is patchy where it occurs (Connor *et al*., 2004). This suggests that the resident species may not be adapted to low oxygen levels but also that deoxygenation of the water column may be short-lived, especially as the biotope is exposed at low tide. Therefore, while some members of the community are known to be tolerant, other species may be lost or reduced in abundance and a resistance of **Medium** is suggested. 'Resilience is probably '**High**' (and sensitivity is assessed as **Low.** However, hypoxia or anoxia caused by the bacterial decomposition of organic matter may be detrimental.

Nutrient enrichment	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	Not sensitive 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 benchmark is set at compliance with WFD criteria for good status, based on nitrogen concentration (UKTAG, 2014).

In-situ primary production is limited to microphytobenthos within and on sediments and the high levels of sediment mobility may limit the level of primary production as abrasion would be likely to damage diatoms (Delgado *et al*., 1991).

Sensitivity assessment. Nutrient level is not a key factor structuring the biotope at the pressure benchmark. In general, however, primary production is low, this biotope is species poor, and characterizing species may be present at low abundances (depending on wave exposure).

Organic enrichment	High Q: High A: High C: High	High Q: High A: High C: High	Not sensitive Q: High A: High C: High

Benthic responses to organic enrichment have been described by Pearson & Rosenberg (1978) and Gray (1981). In general, moderate enrichment increases food supply and increases productivity and abundance. *Nephtys cirrosa* and *Spio spp.* were classified by Borja *et al.* (2000) as being indifferent to enrichment. Dense *Capitella capitata* populations are frequently located in areas with greatly elevated organic content such as areas of sewage disposal and below fish farms and mussel long lines, even though eutrophic sediments are often anoxic and highly sulfidic (Gray, 1979; Tenore, 1977; Warren, 1977; Tenore & Chesney, 1985; Bridges *et al*., 1994; Haskoning, 2006; Callier *et al*., 2007).

Sensitivity assessment. At the benchmark levels, resistance was assessed as '**High'** as the main characterizing species are tolerant of organic enrichment and an input at the pressure benchmark is considered unlikely to lead to gross pollution effects . A resilience of **'High'** is assigned (by default) and the biotope is assessed as **'Not sensitive'.**

Physical Pressures

	Resistance	Resilience	Sensitivity

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

All marine and estuarine habitats and benthic species within them 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 Q: High A: High C: High	Very Low Q: High A: High C: High	High Q: High A: High C: High

A change to natural or artificial hard substratum would remove this sedimentary biotope and the species. If pockets of fine sediment accumulate in pockets within the substrata then these areas may be re-colonised by species associated with this biotope but these pockets of sediment would not be equivalent to the biotope. Recovery will depend on the re-instatement of suitable habitat.

Sensitivity assessment. Based on the loss of suitable habitat, biotope resistance to this pressure is assessed as **'None'**. Resilience is assessed as **'Very low'** as the pressure benchmark refers to a permanent change. Biotope sensitivity is therefore **'High'.**

Physical change (to another sediment type)	None Q: Low A: NR C: NR	High Q: High A: High C: High	Medium Q: Low A: Low C: Low

The benchmark for this pressure refers to a change in one Folk class. The pressure benchmark originally developed by Tillin *et al*., (2010) used the modified Folk triangle developed by Long (2006) which simplified sediment types into four categories: mud and sandy mud, sand and muddy sand, mixed sediments and coarse sediments. The change referred to is, therefore, a change in sediment classification rather than a change in the finer-scale original Folk categories (Folk, 1954). The change in one Folk class is considered to relate to a change in classification to adjacent categories in the modified Folk triangle (Long, 2006). As this biotope occurs within fine sands (JNCC, 2015), the change at the pressure benchmark refers to a potential change to coarse sediments, mixed sediments, sand and muddy sands or mud.

The particle size of sediments and correlated physical and chemical factors (such as drainage, organic matter content and hydrodynamic regime), is a key determinant of the structure of benthic invertebrate assemblages (Van Hoey *et al*., 2004; Yates *et al*., 1993). Infauna can be affected by changes in sediment as many are adapted to burrow through certain grades of sediment (Trueman & Ansell, 1969), decreased fine fractions will reduce habitat suitability for species that maintain permanent burrows. Changes in sedimentary features may also influence the proportions of suspension and deposit feeding animals (Sanders, 1968), with deposit feeders favoured by increases in the proportion of silts and clays. In North America, cultivation of clam species including the Manila clam, *Tapes philippinarum* usually involves some form of habitat modification in the form of adding gravel or gravel and crushed shell over mud and sand beaches, to create a more productive clam habitat (referred to as 'gravelled clam plots'). Such habitat modifications lead to alterations in the local environment and consequently faunal composition. Simenstad and Fresh (1995, cited in Kaiser & Beadman, 2002) reported that the application of gravel to intertidal sediments resulted in a shift from a polychaete to a bivalve and nemertean dominated community, but emphasised that changes are likely to be site-specific.

Sensitivity assessment. Individual members of the community are found in a range of different sediment types, at different abundances. The character of the habitat is largely determined by the sediment type, changes to this would lead to habitat re-classification. The addition of coarse sand particles or fine particles in sufficient quantities would lead to the development of a different habitat type. Changes in sediment characteristics can lead to changes in community structure. An increase in coarse sediments would lead to the development of a community typical of mixed sediments, clean sands and/or gravels depending on the degree of change. In general, an increase to very coarse sediments may favour some amphipod species rather than burrowing polychaetes and sessile tube-dwelling polychaetes. A change to a muddier biotope would lead to reclassification to a biotope within the LS.LSa.MuSa with bivalves such as *Cerastoderma edule* or *Macoma balthica* favoured as these can switch between filter and deposit feeding depending on turbidity from resuspended finer particles. This change would alter the character of the biotope present leading to re-classification, biotope resistance is assessed as **'None'** and, as the change is permanent, resilience is assessed as **'Very Low'.** Biotope sensitivity is therefore '**High'.**

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

The process of extraction is considered to remove all biological components of the biotope group. If extraction occurred across the entire biotope, loss of the biotope would occur. Recovery would require substratum to return to sand .

Sensitivity assessment. Resistance of the biotope to extraction is probably **'None'**. Resilience differs between species with slower recovery likely to be displayed by *Nephtys cirrosa*. Resilience is assessed as '**High**' (although if the substratum changed recovery could be prolonged) and biotope sensitivity is assessed as '**Medium**'.

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

Fine sands will retain water better than coarse sands and will be relatively cohesive and therefore resistant to erosion following surface disturbance. The characterizing bivalve *Angulus tenuis* and other species require contact with the surface for respiration and feeding, fragile animals that are buried close to the surface will be vulnerable to damage, depending on the force of the surface abrasion. Surface compaction can collapse burrows and reduce the pore space between particles, decreasing penetrability and reducing stability and oxygen content. The tops of burrows may be damaged and repaired subsequently at energetic cost to their inhabitants. Experiments with trampling, a pathway for compaction effects, have shown that areas subject to compaction tend to have reduced species abundance and diversity (see trampling pathway below). Sheehan et al. (2007) proposed that following compaction, organisms avoid or emigrate from affected areas.This biotope is present in disturbed and well sorted sands, the associated species are generally present in low abundances and adapted to frequent disturbance. Therefore, resistance to surface abrasion is probably 'High'. The polychaete *Nephtys cirrosa* is adapted to life in unstable sediments and survives through rapid burrowing (McDermott, 1983, cited from Elliott *et al*., 1998). This characteristic is likely to protect this species from surface abrasion.

Paraonis fulgens were found to reduce in abundance in experimental areas exposed to trampling (Reyes-Martínez *et al*., 2015), suggesting a lower resistance of this species to abrasion or surface disturbance. Chandrasekara and Frid (1996) found that some species including *Capitella capitata* and *Scoloplos armiger* reduced in abundance in intertidal muds,along a pathway heavily trampled for five summer months (ca 50 individuals a day Bonsdorff & Pearson (1997) found that sediment disturbance forced *Capitella capitata* deeper into the sediment, although the species was able to burrow back through the sediment to the surface again.Juveniles and adults of *Scoloplos armiger* stay permanently below the sediment surface and freely move without establishing burrows. While juveniles are only found a few millimeters below the sediment surface, adults may retreat to 10 cm depth or more (Reise, 1979; Kruse *et al*., 2004) and are likely to be more protected. The egg cocoons are laid on the surface and hatching time is 2-3 weeks during which these are vulnerable to surface abrasion.

A number of studies have assessed the effects of trampling on other intertidal amphipods and these assessments are used as a proxy. Comparisons between shores with low and high levels of trampling found that the amphipod *Bathyporeia pelagica* is sensitive to human trampling, other species including *Pontocrates arenarius* and the isopod *Eurydice affinis* also decreased in response to trampling but *Bathyporeia pelagica* appeared to be the most sensitive (Reyes-Martínez *et al*., 2015). Changes in abundance of talitrid amphipods on urban beaches subject to high levels of recreational use was also observed by Bessa *et al*. (2014), this study compared abundances between samples taken ten years apart and thus the trends observed were not directly attributable to trampling vs beach cleaning or other pressures although they illustrate a general trend in density patterns as recreational use increases. Ugolini *et al*. (2008) carried out a controlled trampling experiment on *Talitrus saltator*. Plastic cylinders of 110 cm diameter (area 0.95 m^2) were placed in the sand and all individuals trapped and counted, and 400 steps were made in a cylinder in 15 minutes after the amphipods had reburied. The trampling rate was based on observed number of beach users and therefore represents a realistic level of exposure. Alive individuals were counted at the end of the experiment and 24 hours after. Trampling significantly reduced abundance of the amphipods and after 24 hours the percentage of surviving amphipods dropped to almost zero, while survival rates of control (untrampled) amphipods were unaffected. Abrasion and compaction can, therefore, kill buried amphipods within sediments.

Sensitivity assessment. The characterizing species are likely to be reduced following abrasion impacts. However, species in the biotope are adapted to some sediment disturbance. Resistance is assessed as **'Medium'**, and resilience is assessed as '**High**' as migration and water transport of adults will support recolonisation. Sensitivity is, therefore, assessed as '**Low**'.

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

Kaiser et al (2006) found that in sand habitats (grain size not defined by many of the studies included in the analysis), intertidal dredging produced the most severe initial impact out of all of the fishing activities analysed, and no recovery had occurred by the final observation time period (time category > 50 days).

Dernie et al. (2003) compared the recovery rate of benthic assemblages in different sediment types following physical disturbance (the creation of a 'pit' in the sediment surface, the scale of which was chosen to be relevant to bait digging, hand-raking, suction dredging and some forms of trawling) of different intertidal habitats (clean sand (< 3% silt and clay), silty sand (5-20% silt and clay), sandy mud (35-45% silt and clay) and mud (>55% silt and clay)) in the Menai Strait, North Wales.

Ferns et al. (2000) studied the effect of experimental tractor dredging for cockles on non-target invertebrates in areas of both intertidal clean sand and intertidal muddy sand at Burry Inlet, South Wales. The study showed that mechanical cockle harvesting resulted in the loss of a significant proportion of the most common invertebrates from both the clean sand and muddy sand areas. Annelids, molluscs and crustacean declined by 32%, 45% and 81% respectively in the clean sand area respectively post harvesting. Invertebrate populations in the clean sand area with relatively few cockles recovered more quickly than those in the muddy sand area. The time to recovery for the most abundant invertebrate species in the clean sand area were: Tetrastemma sp., reduced by 55% post harvesting, 8 days to recovery; Bathyporeia pilosa, reduced by 82% post harvesting, 39 days to recovery; and Hydrobia ulvae, 56% reduction post harvesting, 8 days to recovery.

Rostron (1995, cited in Gubbay & Knapman 1999) undertook experimental dredging of sandflats with a mechanical cockle dredger, including a site comprising stable, poorly sorted fine sands with small pools and Arenicola marina casts with some algal growths. At this site, post-dredging, there was a decreased number of Pygospio elegans with no recovery to pre-dredging numbers after six months and disappearance of Scoloplos armiger from some dredged plots. The distribution of Nephtys hombergii was disturbed by dredging, with recovery after six months. There was a large decline in numbers of Hydrobia ulvae, with statistical differences between the dredged sites and control sites up to six months post-dredging. Cerastoderma edule numbers were reduced by dredging, with significant reductions in numbers compared with the control still apparent up to six months post-dredging. The dredge tracks were still visible after 6 months (summarised in Gubbay, 1999)

Nephtys cirrosa and Spiophanes bombyx were characterizing species of infauna assemblages in both control and impact sample sites on the Thornton Bank Belgium (North Sea), before and after dredging occurred as part of the construction process for an offshore wind farm (Coates *et al.* 2015). Recovery of assemblages occurred within one to two years at individual dredged sites. The species potentially display resilience to dredging activities as past aggregate dredging had also occurred before wind farm construction.

Nephtys cirrosa was found to be sensitive to experimental trawling disturbance over 18 months (Tuck *et al.*, 1998). *Nephtys cirrosa* is also likely to be vulnerable to dredging but can probably accommodate limited sediment deposition from the dredging process (MES, 2010).

Collie *et al.* (2000) found that abundance of *Nephtys hombergii* was negatively affected by fishing activities. Mean response of infauna and epifauna communities to fishing activities was also much more negative in mud and sand communities (such as this biotope) than other habitats. *Nephtys hombergii* abundance also significantly decreased in areas of the Solent, UK, where bait digging had occurred (Watson *et al.* 2007). Similarly, *Nephtys hombergii* abundance was reduced by 50% in areas where tractor towed cockle harvesting was undertaken on experimental plots in Burry inlet, south Wales, and had not recovered after 86 days (Ferns *et al.*, 2000).

Capitella capitata, are soft bodiedrelatively fragile species inhabiting mucus tubes close to the sediment surface. Abrasion and compaction of the surficial layer may damage individuals. *Capitella capitata* and *Pygospio elegans* were categorised as AMBI fisheries Group IV- as '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' (Gittenberger & Van Loon 2011).

Spio filicornis is a soft bodied organism that exposes its palps at the surface while feeding. It lives infaunally in sandy sediment and any physical disturbance that penetrates the sediment, for example dredging or dragging an anchor, would lead to physical damage of *Spio filicornis*. However, adult worms can burrow up to 10 cm and may escape the disturbance. Juveniles can only burrow up to 2 cm into the sediment and are likely to be affected. However, individuals are likely to pass through a passing scallop dredge due to their small size. Bergman and Hup (1992) reported that the total density of spionids actually increased with increased fishing disturbance presumably due to their ability to colonize newly exposed substratum. Hall *et al.* (1990) investigated the impact of hydraulic dredging for razor clams. They reported that any effects only persisted for a short time, with the community restored after approximately 40 days in stormy conditions. The population density of *Spio filicornis* was slightly reduced in the dredged site relative to the control site but its abundance had increased over that of the control site after 40 days. However, the control site showed a similar level of variation in abundance.

Bergman and Santbrink (2000) found that direct mortality of gammarid amphipods, following a single passage of a beam trawl (in silty sediments where penetration is greater) was 28%. Similar results were reported from experiments s in shallow, wave disturbed areas, using a toothed, clam dredge. *Bathyporeia* spp. experienced a reduction of 25% abundance in samples immediately after intense clam dredging, abundance recovered after 1 day (Constantino *et al*., 2009). Experimental hydraulic dredging for razor clams resulted in no statistically significant differences in *Bathyporeia elegans* abundances between treatments after 1 or 40 days (Hall *et al*., 1990), suggesting that recovery from effects was very rapid. Ferns *et al.* (2000) examined the effects of a tractor-towed cockle harvester on benthic invertebrates and predators in intertidal plots of muddy and clean sand. Harvesting resulted in the loss of a significant proportion of the most common invertebrates from both areas. In the muddy sand, the population of *Bathyporeia pilosa* remained significantly depleted for more than 50 days, whilst the population in clean sand recovered more quickly. These results agree with other experimental studies that clean sands tend to recover more quickly that other habitat types with higher proportions of fine sediment (Dernie *et al*., 2003).

Bergman and Hup (1992) found that worm species (including *Scoloplos armiger*) showed no change in total density after trawling a subtidal habitat. Conversely, a later study by Bergman and Santbrink (2000) found that the direct mortality of *Scoloplos armiger* from a single passage of a beam trawl in subtidal silty grounds was 18% of the population. Rostron (1995) undertook experimental dredging of sandflats with a mechanical cockle dredger, including a site comprised of stable, poorly sorted fine sands with small pools and *Arenicola marina* casts with some algal growths. At this site, post-dredging *Scoloplos armiger* had disappeared from some dredged plots. Ferns*et al*. (2000) used a tractor-towed cockle harvester, to extract cockles from intertidal plots of muddy sand and clean sand, to investigate the effects on non-target organisms; 31% of the population of *Scoloplos armiger* (initial density of 120 per m2) were removed. Populations of *Scoloplos armiger* remained significantly depleted in the area of muddy sand for more than 50 days after harvesting. Ball *et al*. (2000) found that species including*Scoloplos armiger* showed a significant decrease in abundance of between 56-27% after 16 months of otter trawling at a previously unfished Scottish sea loch. Chandrasekara and Frid (1996, cited in Tyler-Walters & Arnold, 2008) found that along a pathway heavily used for five summer months (ca. 50 individuals day-1), *Scoloplos armiger* reduced in abundance. Recovery took place within 5-6 months. These studies suggest that *Scoloplos armiger*is likely to be impacted by sediment disturbance and that recovery to previous densities may require more than two years.

A number of studies have found that the abundance of the polychaete *Pygospio elegans* is reduced by simulated cockle dredging (Hall & Harding, 1998; Moore, 1990; Ferns *et al*., 2000; Rostron, 1995). Ferns *et al*. (2000) found that tractor towed cockle harvesting removed 83% of *Pygospio elegans* (initial density 1850/ m2). In muddy sand habitats, *Pygospio elegans* had not recovered to the original abundance after 174 days (Ferns *et al*.,2000). Rostron (1995) also found that *Pygospio elegans* had not recovered to pre-dredging numbers after six months. Conversely, Hall & Harding, (1998) found that abundance of *Pygospio elegans* increased significantly over 56 days following suction dredging. *Pygospio elegans* inhabits a fragile tube that projects above the sediment surface and is probably more vulnerable to physical disturbance and abrasion than other, more deeply buried, infaunal species.

Sensitivity assessment. Although some polychaetes may be able to reposition following sedimentation at the pressure benchmark this will depend on the characteristics of the overburden and sedentary species such as *Pygospio elegans* are likely to suffer high levels of mortality. Resistance of the biotope is assessed as '**Low**', as a proportion of the population of characterizing species may be removed, however, species in the biotope are adapted to disturbance and recover quickly Hence, resilience is assessed as '**High** ', and sensitivity is assessed as '**Low**'.

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

The characterizing species live within the sand and are unlikely to be directly affected by an increased concentration of suspended matter in the water column. An increase in organic particles may benefit the characterizing species *Macoangulus tenuis*Within the mobile sands habitat storm events or spring tides may re-suspend or transport large amounts of material and therefore species are considered to be adapted to varying levels of suspended solids. which is a suspension feeder. *Bathyporeia* spp. feed on diatoms within the sand grains (Nicolaisen & Kanneworff, 1969), an increase in suspended solids that reduced light penetration could alter food supply. However, diatoms are able to photosynthesize while the tide is out and therefore a reduction in light during tidal inundation may not affect this food source, depending on the timing of the tidal cycle. *Bathyporeia* spp. may be regular swimmers within the surf plankton, where the concentration of suspended particles would be expected to be higher (Fincham, 1970a).

However, the biotope is characterized by a low amount of organic matter and an increase in suspended solids may cause a change in this factor if this is coupled with changes in hydrodynamics that reduce particle re-suspension. Increased suspended solids are unlikely to have a direct impact on infauna but increased organic matter may result in an increase in the abundance of opportunistic species such as *Capitella capitella* and oligochaetes. Biotope resistance is assessed as '**High**' and resilience as '**High**' (by default), so that the biotope is assessed to be '**Not sensitive**'.

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

Mobile and/or burrowing species (including molluscs and polychaetes such as *Nephtys* spp., and *Scoloplos armiger*) are generally considered to be able to reposition following periodic siltation events or low levels of chronic siltation. Venerid bivalves are typically able to relocate within the sediment in response to siltation and *Macoangulus tenuis* may be able to relocate to the surface and/or extend its siphons to the surface to maintain respiration while burrowing. *Nepthys cirrosa* occurs in fine to coarser sands, with greatest abundance in the Belgium part of the North Sea recorded in medium grain sizes (Degraer *et al*., 2006). A light deposition of fine sediment may lead to small but insignificant changes in abundance as it will reduce the available preferred habitat with medium grain size. As the tidal flow is likely to be relatively strong in this biotope, a light deposition of finer sediment is likely to be resuspended.

Sensitivity assessment. None of the characterizing species are considered likely to be significantly impacted by deposition of up to 5 cm of fine material. Resistance is assessed as '**High'**. Resilience as '**High'** and Sensitivity as '**Not sensitive'**.

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

Studies have found that beach 'replenishment' or 'nourishment' that involves the addition of sediments on beaches can have a number of impacts on the infauna (Peterson *et al*., 2000; Peterson *et al*., 2006). Impacts are more severe when the sediment added differs significantly in grain size or organic content (Nelson *et al*., 1989; Peterson *et al*., 2000). For example, Maurer *et al*. (1981) found that the amphipod *Parahaustorius longimerus* which occurs intertidally in clean, well-sorted sands and is an active, effective burrower was able to regain the surface after being buried by sand far more easily than when buried under silt/clay mixtures.

Bijkerk (1988, results cited from Essink, 1999) indicated that the maximal overburden through which small bivalves could migrate was 20 cm in sand for *Donax* and approximately 40 cm in mud for *Tellina* sp. and approximately 50 cm in sand. No further information was available on the rates of survivorship or the time taken to reach the surface.

Nephtys cirrosa is a large infaunal species, with adult size between 6 cm and 10 cm and capable of moving through the sediment, suggesting some resilience to smothering. *Nephtys cirrosa* is an active worm which demonstrates the characteristic swimming motion (a rapid lateral wriggling, starting from the rear and increasing in amplitude towards the head) of the Nephtyidae. Deposition of up to 30 cm of fine material is likely to bury some individuals beyond the typical 5 to 15 cm depth of tunnels. It is likely *Nephtys cirrosa* close to the surface may be capable of relocating in the sediment although feeding and reproduction activities are likely to be interrupted.

Nepthys cirrosa occurs in fine to coarser sands, with greatest abundance in the Belgium part of the North Sea recorded in medium grain sizes (Degraer *et al*., 2006). Presence of fine material may lead to small but insignificant changes in abundance as it will reduce the available preferred habitat with medium grain size. As the tidal flow is strong in this biotope, a light deposition of finer sediment is likely to be resuspended. Resistance is likely to be high to the presence of finer material for *Nepthys cirrosa* but initial smothering is likely to cause some mortality and interrupt feeding and reproduction activity at the benchmark level.

Bijkerk (1988, results cited from Essink, 1999) found that the maximal overburden through which *Bathyporeia* could migrate was approximately 20 cm in mud and 40 cm in sand. No further information was available on the rates of survivorship or the time taken to reach the surface.

Sensitivity assessment. Overall smothering by 30 cm of fine sediments may result in mortality of characterizing species. The introduction of fine sediment may also alter the sediment typical of the biotope causing a temporary shift in the abundance of species. However, the opportunistic species occurring in the biotope are likely to recover rapidly following sediment recovery. Biotope resistance is, therefore, assessed as **'Low'**, resilience is assessed as **'High',** following habitat recovery to fine sands and biotope sensitivity is assessed as '**Low**'.

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

Plastic debris breaks up to form microplastics. Microplastics have been shown to occur in marine sediments and to be ingested by detritivores such as the amphipod *Orchestia gammarellus*, deposit feeders such as *Arenicola marina* and holothurians, as well as by suspension feeders, e.g. *Mytilus edulis* (Wright *et al*., 2013b; Browne *et al*., 2015).

Wright *et al.* (2013) showed that the presence of microplastics (5% UPVC) in a lab study significantly reduced feeding activity when compared to concentrations of 1% UPVC and controls. As a result, *Arenicola marina* showed significantly decreased energy reserves (by 50%), took longer to digest food, and as a result decreased bioturbation levels, which would be likely to impact colonization of sediment by other species, reducing diversity in the biotopes the species occurs within. Wright *et al.* (2013) suggested that in the intertidal regions of the Wadden Sea, where *Arenicola marina* is an important ecosystem engineer, *Arenicola marina* could ingest $33m^3$ of microplastics a year.

In a similar experiment, Browne *et al.* (2013) exposed *Arenicola marina* to sediments with 5% PVC particles or sand presorbed with pollutants nonylophenol and phenanthrene for 10 days. PVC is dense and sinks to the sediment. The experiment used Both microplastics and sand transferred the pollutants into the tissues of the lugworm by absorption through the gut. The worms accumulated over 250% more of these pollutants from sand than from the PVC particulates. The lugworms were also exposed to PVC particulates presorbed with plastic additive, the flame retardant PBDE-47 and antimicrobial Triclosan. The worms accumulated up to 3,500% of the concentration of theses contaminants when compared when to the experimental sediment. Clean sand and PVC with contaminants reduced feeding but PVC with Triclosan reduced feeding by over 65%. In the PVC with Triclosan treatments, 55% of the lugworms died. Browne *et al*., 2013 concluded that the contaminants tested reduced feeding, immunity, response to oxidative stress, and survival (in the case of Triclosan).

Sensitivity assessment. Impacts from the pressure 'litter' would depend on upon the exact form of litter or man-made object being introduced. Browne *et al.* (2015) suggested that if effects in the laboratory occurred in nature, they could lead to significant changes in sedimentary communities as *Arenicola marina* is an important bioturbator and ecosystem engineer in sedimentary habitats. *Arenicola marina* does not reach high abundances in this biotope but other deposit feeding polychaetes could potentially ingest microplastics, although no evidence in available at present. This pressure is **'Not assessed'** as no benchmark has been defined for this pressure.

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

Electric and magnetic fields generated by sources such as marine renewable energy device/array cables may alter behaviour of predators and affect infauna populations. Evidence is limited and occurs for electric and magnetic fields below the benchmark levels, confidence in evidence of these effects is very low.

Field measurements of electric fields at North Hoyle wind farm, North Wales recorded 110μ V/m (Gill *et al.* 2009). Modelled results of magnetic fields from typical subsea electrical cables, such as those used in the renewable energy industry produced magnetic fields of between 7.85 and 20 μT (Gill *et al.* 2009; Normandeau *et al.* 2012). Electric and magnetic fields smaller than those recorded by in field measurements or modelled results were shown to create increased movement in thornback ray *Raja clavata* and attraction to the source in catshark *Scyliorhinus canicular* (Gill *et al.* 2009).

Flatfish including dab *Limanda limanda* and sole *Solea solea* are predators of many polychaete species. They have been shown to decrease in abundance in a wind farm array or remain at distance from wind farm towers (Vandendriessche *et al.*, 2015; Winter *et al.* 2010). However, larger plaice increased in abundance (Vandendriessche *et al.*, 2015). There have been no direct causal links identified to explain these results.

Sensitivity assessment. No evidence was found on effects of electric and magnetic fields on the characterizing species. However, responses by flatfish and elasmobranchs suggest changes in predator behaviour are possible. There is no evidence currently but if electromagnetic fields affect predator-prey dynamics as further marine renewable energy devices are deployed, these are likely to be over small spatial scales and unlikely to significantly impact the biotope.

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

Species within the biotope can probably detect vibrations caused by noise. However, at the benchmark level the community is unlikely to be sensitive to noise and this pressure is therefore '**Not relevant'.**

Introduction of light or shading	High Q: Low A: NR C: NR	High Q: High A: High C: High	Not sensitive Q: Low A: Low C: Low

As this feature is not characterized by the presence of primary producers it is not considered that shading would alter the character of the habitat. As the characterizing biological assemblage occurs within the sediment, an increase in light or shading is considered 'Not relevant'. However, shading may reduce the microphytobenthos component of this infralittoral biotope. Mucilaginous secretions produced by these algae may stabilize fine substrata (Tait & Dipper, 1998). Shading will prevent photosynthesis leading to death or migration of sediment microalgae, which may alter sediment cohesion and food supply to higher trophic levels.

Sensitivity assessment. Changes in light are not considered to directly affect the biotope. However, some changes in behaviour or food supply for characterizing species could result. Overall, this Resistance is assessed as **High**. Therefore, resilience is **High** and the biotope is assessed as **Not sensitive**

Barrier to species movement	High Q: Low A: NR C: NR	High Q: High A: High C: High	Not sensitive Q: Low A: Low C: Low

The characterizing species *Macoangulus tenuis*, *Spio filicornis* and *Nephtys cirrosa* produce pelagic larvae. Barriers that limit tidal excursion and flushing may reduce connectivity but equally may help to retain larvae within areas of suitable habitat. *Pygospio elegans* are capable of both benthic and pelagic dispersal and *Bathyporeia* spp brood young. In the sheltered waters where some examples of this biotope occur, *in-situ* reproduction may maintain populations rather than long-range pelagic dispersal. The biotope is considered to have '**High'** resistance to the presence of barriers that lead to a reduction in tidal excursion, resilience is assessed as '**High**' (by default) and the biotope is considered to be '**Not sensitive'**

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

Characterizing species may have some, limited, visual perception. As they live in the sediment the species will most probably not be impacted at the pressure benchmark and this pressure is considered **'Not relevant'.**

Biological Pressures

	Resistance	Resilience	Sensitivity
Genetic modification & translocation of indigenous species	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR

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

Coastal and estuarine areas are among the most biologically invaded systems in the world, especially by molluscs such as the slipper limpet *Crepidula fornicata* and the Pacific oyster *Magallana gigas* (OSPAR, 2009b). The two species have not only attained considerable biomasses from Scandinavian to Mediterranean countries but have also generated ecological consequences such as alterations of benthic habitats and communities, or food chain changes. In the Wadden Sea, the main issue of concern is the Pacific oyster (*Magallana gigas*), which has also spread in the Thames estuary and along French intertidal flats. Padilla (2010) predicted that *Magallana gigas* could either displace or overgrown mussels on rocky and sedimentary habitats of low or high energy. In general littoral sand sediments are mobile and winter storms may remove sediments and wash-out some species (Connor *et al*., 2004) preventing the establishment of larger, longer-lived species and the development of bivalve reefs. However, as some beaches in which the biotope occur may be relatively sheltered some colonization may occur and sensitivity to invasive molluscs is considered.

In the Wadden Sea and the North Sea, *Magallana gigas* overgrows mussel beds in the intertidal zone (Diederich 2005, 2006; Kochmann *et al.*, 2008), although they did show a preference for settling on conspecifics before the mussels and struggled to settle on mussels with a fucoid covering. However, recruitment of *Magallana gigas* was significantly higher in the intertidal than the shallow subtidal, although the survival of adult oysters or mussels in the subtidal is limited by predation.

Crepidula fornicata is known to colonize and smother a wide range of sediments in the subtidal, from mixed sediments to mud, especially in prior shellfish beds (e.g. of oysters and mussels) (Blanchard, 1997; Minchin *et al.*, 1995). *Crepidula fornicata* larvae may out-compete oyster (*Magallana gigas*) larvae during summer months where the two species co-occur. Trophic competition between adult *Crepidula fornicata* and *Magallana gigas* was reported in France during winter and spring. In Mont Saint-Michel Bay, France, slipper limpet populations have affected flatfish populations. Changes in habitat structure and reduced abundance of suspension feeding organisms upon which the flatfish feed were linked to slipper limpet extent (Decottignies *et al.*, 2007; Blanchard *et al.* 2008; and Kostecki *et al.*, 2011 cited in Sewell & Sweet, 2011).

On some north Wales shores *Ensis siliqua* occurs (Connor *et al.*, 2004), this species could co-occur with or be replaced by a similar, but non-native species *Ensis directus*. Such a change is unlikely to alter the character of the biotope.

Sensitivity assessment. *Magallana gigas* is predicted to invade sedimentary habitats, although no direct examples exist to date and *Magallana gigas* recruitment is lower in the subtidal (Diederich 2005, 2006; Padilla, 2010). *Crepdiula fornicata* is a major invader and colonizer of subtidal sediments. However, both species require hard substrata in the form of stones, debris or, preferably, the shells conspecifics to colonize the habitat. This biotope is dominated by fine sand and a shell fraction is not recorded in the description (Connor *et al.*, 2004) but if artificial hard debris (e.g. litter) was introduced to the habitat then it may provide an initial point for the colonization of *Crepidula* in particular. Although it would probably take many years, colonization by *Crepidula* would result in the complete modification of the habitat, reclassification and loss of the biotope, although polychaete populations may survive in the sediment itself. Therefore, a precautionary resistance of **Low** has been suggested with 'Low' confidence due to the lack of direct evidence. Resilience is likely to be **Very low** as a bed of *Crepidula* or *Magallana gigas* would need to be removed before recovery could begin. Therefore, sensitivity is assessed as **High**.

Introduction of microbial pathogens	No evidence (NEv) Q: NR A: NR C: NR	No evidence (NEv) Q: NR A: NR C: NR	No evidence (NEv) Q: NR A: NR C: NR

No evidence was found to assess this pressure.

Removal of target species	Low Q: High A: High C: High	High Q: High A: Medium C: High	Medium Q: High A: Medium C: High

The only species characterizing the biotope that is likely to be removed is *Nephtys cirrosa which* is targeted by bait diggers. There is limited information on the effect of targeted removal on *Nephtys* *cirrosa* populations, however, there is evidence on effects on *Nephtys hombergii*. *Nephtys hombergii* is directly removed through commercial bait digging and by recreational anglers and abundance significantly decreased in areas of the Solent, UK, where bait digging (primarily for *Nereis virens*) had occurred (Watson *et al.* 2007). Recovery of *Nephtys hombergii* has been assessed to be high as re-population would occur initially relatively rapidly via adult migration and later by larval recruitment. Dittman *et al*. (1999) observed that *Nephtys hombergii* was amongst the macrofauna that colonized experimentally disturbed tidal flats within two weeks of the disturbance that caused defaunation of the sediment. However, if sediment is damaged recovery is likely to be slower, for instance, *Nephtys hombergii* abundance was reduced by 50% in areas where tractor towed cockle harvesting was undertaken on experimental plots in Burry inlet, south Wales, and had not recovered after 86 days (Ferns *et al*., 2000).

Removal of *Nephtys cirrosa* by bait digging may cause short-term loss of food resources for predators such as fish species including *Limanda limanda* and *Pleuronectes platessa.* As recovery is medium to high the long term impacts on populations are likely to be small, but will be dependent upon the scale and frequency of bait digging activities.

Sensitivity assessment. Confidence in this assessment in relation to the removal of *Nephtys cirrosa* is low as it is based on evidence of removal of *Nephtys hombergii*. However, biotope resistance is assessed as '**Low**' based on direct removal of a characterizing species, Resilience is assessed as **'High'** as habitats that are not regularly harvested may recover rapidly, it should be noted that continued harvesting will inhibit recovery. Biotope sensitivity to a single harvesting event is assessed as **'Low'**. It is important to consider that the spatial extent and duration of harvesting is important to consider when assessing this pressure as smaller scale extraction may not impact the entire extent of the biotope but greater scale extraction over a long period would cause longer-term impacts.

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

Direct, physical impacts are assessed through the abrasion and penetration of the seabed pressures, while this pressure considers the ecological or biological effects of by-catch. Species in this biotope, including the characterizing species, may be damaged or directly removed by static or mobile gears that are targeting other species (see abrasion and penetration pressures). A study of the effects of hydraulic dredging in Dundalk Bay, Ireland indicated that there was a short-lived effect (<4 months) of the fishery on *Macoangulus* (studies as *Angulus) tenuis* and the target species *Cerastoderma edule*, which spatially overlapped (Clarke & Tully, 2011). The authors concluded that the dominant species in the benthic community, including *Macoangulus* (as *Angulus) tenuis*, and a number of polychaete species had low sensitivity (high resilience and high recoverability) to disturbance.

Collie *et al*. (2000) found that abundance of a *Nephtys hombergii* was negatively affected by fishing activities. Mean response of infauna and epifauna communities to fishing activities was also much more negative in mud and sand communities (such as this biotope) than other habitats. *Nephtys hombergii* abundance also significantly decreased in areas of the Solent, UK, where bait digging had occurred (Watson *et al.* 2007). Similarly, *Nephtys hombergii* abundance was reduced by 50% in areas where tractor towed cockle harvesting was undertaken on experimental plots in Burry inlet, south Wales, and had not recovered after 86 days (Ferns *et al*., 2000).

Sensitivity assessment. The incidental damage or removal of a proportion of the population (e.g.

by commercial bait digging) may change the character of the community temporarily. The biotope is disturbed seasonally by storms, (Connor *et al*., 2004) and is likely to recover quickly. However, disturbance from repeated events e.g. by periodic bait digging (see above) may prolong recovery. Biotope resistance is assessed as '**Low**' based on removal or damage of characterizing species, that on commercial scales can remove a large proportion of the population and lead to an impacted community. Resilience is assessed as **'High'** but it should be noted that continued harvesting will impact the population and *Nephtys cirrosa* will take longer to recover if harvesting is over extended spatial scales.Biotope sensitivity is assessed as **'Low'**. It is important to consider that the spatial extent and duration of areas impacted by removal is important to consider when assessing this pressure, as smaller scale extraction may not impact the entire extent of the biotope but greater scale extraction over a long period would cause longer term impacts. The type of fishing activity is also important to consider in relation to the type and severity of the impact.

Bibliography

Alheit, J., 1978. Distribution of the polychaete genus *Nephtys*: a stratified random sampling survey. *Kieler Meeresforschungen*, **4**, 61-67.

Arndt, C. & Schiedek, D., 1997. *Nephtys hombergii*, a free living predator in marine sediments: energy production under environmental stress. *Marine Biology*, **129**, 643-540.

Bergman, M.J.N. & Hup, M., 1992. Direct effects of beam trawling on macro-fauna in a sandy sediment in the southern North Sea. *ICES Journal of Marine Science*, **49**, 5-11.

Beukema, J.J., De Bruin, W. & Jansen, J.J.M., 1978. Biomass and species richness of the macrobenthic animals living on the tidal flats of the Dutch Wadden Sea: Long-term changes during a period of mild winters. *Netherlands Journal of Sea Research*, **12**, 58-77.

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

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.

Bolam, S.G. & Fernandes, T.F., 2002. Dense aggregations of tube-building polychaetes: response to small-scale disturbances. *Journal of Experimental Marine Biology and Ecology*, **269**, 197-222.

Bonsdorff, E. & Pearson, T.H., 1997. The relative impact of physical disturbance and predation by *Crangon crangon* on population density in *Capitella capitella*: An experimental study. *Ophelia*, **46**, 1-10.

Boon, J.P., Zantvoort, M.B., Govaert, M.J.M.A. & Duinker, J.C ., 1985. Organochlorines in benthic polychaetes (*Nephtys* spp.) and sediments from the southern North Sea. Identification of individual PCB components. *Netherlands Journal of Sea Research*, **19**, 93-109.

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.

Bridges, T.S., Levin, L.A., Cabrera, D. & Plaia, G., 1994. Effects of sediment amended with sewage, algae, or hydrocarbons on growth and reproduction in two opportunistic polychaetes. *Journal of Experimental Marine Biology and Ecology*, **177** (1), 99-119.

Browne, M.A., Underwood, A.J., Chapman, M.G., Williams, R., Thompson, R.C. & van Franeker, J.A., 2015. Linking effects of anthropogenic debris to ecological impacts. Proceedings of the Royal Society B: Biological Sciences, 282 (1807), 20142929.

Browne, Mark A., Niven, Stewart J., Galloway, Tamara S., Rowland, Steve J. & Thompson, Richard C., 2013. Microplastic Moves Pollutants and Additives to Worms, Reducing Functions Linked to Health and Biodiversity. *Current Biology*, **23**(23), 2388-2392.

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.]

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.

Callier, M. D., McKindsey, C.W. & Desrosiers, G., 2007. Multi-scale spatial variations in benthic sediment geochemistry and macrofaunal communities under a suspended mussel culture. *Marine Ecology Progress Series*, **348,** 103-115.

Chandrasekara, W.U. & Frid, C.L.J., 1996. Effects of human trampling on tidal flat infauna. *Aquatic Conservation: Marine and Freshwater Ecosystems*, **6**, 299-311.

Coates, D.A., van Hoey, G., Colson, L., Vincx, M. & Vanaverbeke, J., 2015. Rapid macrobenthic recovery after dredging activities in an offshore wind farm in the Belgian part of the North Sea. *Hydrobiologia*, **756** (1), 3-18.

Collie, J.S., Hall, S.J., Kaiser, M.J. & Poiner, I.R., 2000. A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, **69** (5), 785–798.

Connor, D.W., Allen, J.H., Golding, N., Howell, K.L., Lieberknecht, L.M., Northen, K.O. & Reker, J.B., 2004. The Marine Habitat Classification for Britain and Ireland. Version 04.05. ISBN 1 861 07561 8. In JNCC (2015), *The Marine Habitat Classification for Britain and Ireland Version 15.03.* [2019-07-24]. Joint Nature Conservation Committee, Peterborough. Available from https://mhc.jncc.gov.uk/

Crisp, D.J. (ed.), 1964. The effects of the severe winter of 1962-63 on marine life in Britain. *Journal of Animal Ecology*, **33**, 165-210.

Dales, R.P., 1958. Survival of anaerobic periods by two intertidal polychaetes, *Arenicola marina* (L.) and *Owenia fusiformis* Delle Chiaje. *Journal of the Marine Biological Association of the United Kingdom*, **37**, 521-529.

Degraer, S., Mouton, I., De Neve, L. & Vincx, M., 1999. Community structure and intertidal zonation of the macrobenthos on a macrotidal, ultra-dissipative sandy beach: summer-winter comparison. *Estuaries*, **22**, 742-752.

Degraer, S., Wittoeck, J., Appeltans, W., Cooreman, K., Deprez, T., Hillewaert, H., Hostens, K., Mees, J., Berge, V. & Vincx, M., 2006. *The macrobenthos atlas of the Belgian part of the North Sea.* Belgian Science Policy.

Delgado, M., De Jonge, V. & Peletier, H., 1991. Effect of sand movement on the growth of benthic diatoms. *Journal of Experimental Marine Biology and Ecology*, **145** (2), 221-231.

Desprez, M.H., Rybarczyk, H., Wilson, J.G., Ducrotoy, J.P., Sueur, F., Olivesi, R. & Elkaim, B., 1992. Biological impact of eutrophication in the Bay of Somme and the induction and impact of anoxia. *Netherlands Journal of Sea Research*, **30**, 149-159.

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.

Diederich, S., 2005. Differential recruitment of introduced Pacific oysters and native mussels at the North Sea coast: coexistence possible? *Journal of Sea Research*, **53** (4), 269-281.

Diederich, S., 2006. High survival and growth rates of introduced Pacific oysters may cause restrictions on habitat use by native mussels in the Wadden Sea. *Journal of Experimental Marine Biology and Ecology*, **328** (2), 211-227.

Dittmann, S., Günther, C-P. & Schleier, U., 1999. Recolonization of tidal flats after disturbance. In *The Wadden Sea ecosystem: stability, properties and mechanisms* (ed. S. Dittmann), pp.175-192. Berlin: Springer-Verlag.

Elliot, M., Nedwell, S., Jones, N.V., Read, S.J., Cutts, N.D. & Hemingway, K.L., 1998. Intertidal sand and mudflats & subtidal mobile sandbanks (Vol. II). An overview of dynamic and sensitivity for conservation management of marine SACs. *Prepared by the Scottish Association for Marine Science for the UK Marine SACs Project.*

Emery, K.O. & Stevenson, R.E., 1957. *Estuaries and lagoons.* In *Treatise on marine ecology and paleoecology*.1. *Ecology*, (ed. J.W. Hedgpeth), USA: Geological Society of America.

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

Fallesen, G. & Jørgensen, H.M., 1991. Distribution of *Nephtys hombergii* and *Nephtys ciliata* (Polychaeta: Nephtyidae) in Århus Bay, Denmark, with emphasis on the severe oxygen deficiency. *Ophelia*, Supplement **5**, 443-450.

Ferns, P.N., Rostron, D.M. & Siman, H.Y., 2000. Effects of mechanical cockle harvesting on intertidal communities. *Journal of Applied Ecology*, **37**, 464-474.

Fish, J.D. & Fish, S., 1978. Observations on an annual migration of *Bathyporeia pelagica* (Amphipoda, Haustoriidae). *Crustaceana*, **35**, 215-221.

Fish, J.D. & Preece, G.S., 1970. The annual reproductive patterns of *Bathyporeia pilosa* and *Bathyporeia pelagica* (Crustacea: Amphipoda). *Journal of the Marine Biological Association of the United Kingdom*, **50**, 475-488.

Forbes, T.L. & Lopez, G.R., 1990. The effect of food concentration, body size, and environmental oxygen tension on the growth of the deposit-feeding polychaete, *Capitella* species 1. Limnology and Oceanography, **35** (7), 1535-1544.

Gamenick, I. & Giere, O., 1997. Ecophysiological studies on the *Capitella capitata* complex: respiration and sulfide exposure. *Bulletin of Marine Science*, **60**, 613.

Gaston, G.R., McLelland, J.A. & Heard, R.W., 1992. Feeding biology, distribution, and ecology of two species of benthic polychaetes: *Paraonis fulgens* and *Paraonis pygoenigmatica* (Polychaeta: Paraonidae). *Gulf Research Reports*, **8** (4), 395-399.

Giere, O. & Pfannkuche, O., 1982. Biology and ecology of marine Oligochaeta, a review. *Oceanography and Marine Biology*, **20**, 173-309.

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.

Gill, A.B., Huang, Y., Gloyne-Philips, I., Metcalfe, J., Quayle, V., Spencer, J. & Wearmouth, V., 2009. COWRIE 2.0 Electromagnetic Fields (EMF) Phase 2: EMF-sensitive fish response to EM emissions from sub-sea electricity cables of the type used by the offshore renewable energy industry. *Commissioned by COWRIE Ltd (project reference COWRIE-EMF-1-06)*, **68**.

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

Grassle, J.F. & Grassle, J.P., 1974. Opportunistic life histories and genetic systems in marine benthic polychaetes. *Journal of Marine Research*, **32**, 253-284.

Grassle, J.F. & Grassle, J.P., 1976. Sibling species in the marine pollution indicator (*Capitella*) (Polychaeta). *Science*, **192**, 567-569.

Gray, J.S., 1979. Pollution-induced changes in populations. *Philosophical Transactions of the Royal Society of London*, Series B, **286**, 545-561.

Gray, J.S., 1981. *The ecology of marine sediments. An introduction to the structure and function of benthic communities.* Cambridge: Cambridge University Press.

Hall, S.J., Basford, D.J. & Robertson, M.R., 1990. The impact of hydraulic dredging for razor clams *Ensis* spp. on an infaunal community. *Netherlands Journal of Sea Research*, **27**, 119-125.

Haskoning UK Ltd. 2006. Investigation into the impact of marine fish farm deposition on maerl beds. *Scottish Natural Heritage Commissioned Report No.* 213 (ROAME No. AHLA10020348).

Hayward, P.J. 1994. *Animals of sandy shores.* Slough, England: The Richmond Publishing Co. Ltd. [Naturalists' Handbook 21.]

Hayward, P.J. & Ryland, J.S. (ed.) 1995a. *The marine fauna of the British Isles and north-west Europe. Volume 2. Molluscs to Chordates.* Oxford Science Publications. Oxford: Clarendon Press.

Henriksson, R., 1969. Influence of pollution on the bottom fauna of the Sound (Öresund). *Oikos*, **20** (2), 507-523.

Huthnance, J., 2010. Ocean Processes Feeder Report. London, *DEFRA on behalf of the United Kingdom Marine Monitoring and*

Assessment Strategy (UKMMAS) Community.

JNCC, 2015. The Marine Habitat Classification for Britain and Ireland Version 15.03. (20/05/2015). Available from https://mhc.jncc.gov.uk/

JNCC, 2015. The Marine Habitat Classification for Britain and Ireland Version 15.03. (20/05/2015). Available from https://mhc.jncc.gov.uk/

Junoy, J., Castellanos, C., Vieitez, J.M. & Riera, R., 2013. Seven years of macroinfauna monitoring at Ladeira beach (Corrubedo Bay, NW Spain) after the Prestige oil spill. *Oceanologia*, **55** (2), 393-407.

Kochmann, J., Buschbaum, C., Volkenborn, N. & Reise, K., 2008. Shift from native mussels to alien oysters: differential effects of ecosystem engineers. *Journal of Experimental Marine Biology and Ecology*, **364** (1), 1-10.

Le Moal, Y., 1980. Ecological survey of an intertidal settlement living on a soft substrata in the Aber Benoit and Aber Wrac'h estuaries, after the *Amoco Cadiz* oil spill. Universite de Bretagne Occidentale, Brest (France), 25pp.

Mangum, C. & Van Winkle, W., 1973. Responses of aquatic invertebrates to declining oxygen conditions. *American Zoologist*, **13** (2), 529-541.

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

McLusky, D.S., Anderson, F.E. & Wolfe-Murphy, S., 1983. Distribution and population recovery of *Arenicola marina* and other benthic fauna after bait digging. *Marine Ecology Progress Series*, **11**, 173-179.

Meißner, K., Darr, A. & Rachor, E., 2008. Development of habitat models for *Nephtys* species (Polychaeta: Nephtyidae) in the German Bight (North Sea). *Journal of Sea Research*, **60** (4), 276-291.

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

Mettam, C., 1989. The life cycle of *Bathyporeia pilosa* Lindström (Amphipoda) in a stressful, low salinity environment. *Scientia Marina*, **53**, 543-550.

Mills, D.J.L., 1998. Liverpool Bay to the Solway (Rhos-on-Sea to the Mull of Galloway)(MNCR Sector 11). In *Marine Nature Conservation Review. Benthic marine ecosystems of Great Britain and the north-east Atlantic*, pp. 315-338.

Minchin, D., McGrath, D. & Duggan, C.B., 1995. The slipper limpet *Crepidula fornicata* (L.) in Irish waters with a review of its occurrence in the north east Atlantic. *Journal of Conchology*, **35**, 247-254.

Moore, J., 1991. Studies on the Impact of Hydraulic Cockle Dredging on Intertidal Sediment Flat Communities. *A report to the Nature Conservancy Council from the Field Studies Council Research Centre, Pembroke, Wales*, FSC/RC/4/91.

Nehmer, P. & Kroencke, I., 2003. Macrofaunal communities in the Wichter Ee, a channel system in the East Frisian Wadden Sea. *Senckenbergiana Maritima*, **32** (1-2), 1-10.

Nehmer, P. & Kroencke, I., 2003. Macrofaunal communities in the Wichter Ee, a channel system in the East Frisian Wadden Sea. *Senckenbergiana Maritima*, **32** (1-2), 1-10.

Newell, R.C., Seiderer, L.J. & Hitchcock, D.R., 1998. The impact of dredging works in coastal waters: a review of the sensitivity to disturbance and subsequent biological recovery of biological resources on the sea bed. *Oceanography and Marine Biology: an Annual Review*, **36**, 127-178.

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**, 215-226.

Olsgard, F., 1999. Effects of copper contamination on recolonisation of subtidal marine soft sediments - an experimental field study. *Marine Pollution Bulletin*, **38**, 448-462.

OSPAR, 2009b. Background document for Intertidal mudflats. OSPAR Commission, Biodiversity Series, OSPAR Commission, London, 29 pp. http://www.ospar.org/documents?v=7186

Padilla, D.K., 2010. Context-dependent impacts of a non-native ecosystem engineer, the Pacific Oyster *Crassostrea gigas*. *Integrative and Comparative Biology*, **50** (2), 213-225.

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.

Petrich, S.M. & Reish, D.J., 1979. Effects of aluminium and nickel on survival and reproduction in polychaetous annelids. *Bulletin of Environmental Contamination and Toxicology*, **23**, 698-702.

Preece, G.S., 1971. The swimming rhythm of *Bathyporeia pilosa* (Crustacea: Amphipoda). *Journal of the Marine Biological Association of the United Kingdom*, **51**, 777-791.

Redman, C.M., 1985. Effect of temperature and salinity on the life history of *Capitella capitata* (type I). *Dissertation Abstracts*, **46**, 91.

Reise, K., 1982. Long-term changes in the macrobenthic invertebrate fauna of the Wadden Sea: are polychaetes about to take over?. *Netherlands journal of sea research*, **16**, 29-36.

Reyes-Martínez, M.J., Ruíz-Delgado, M.C., Sánchez-Moyano, J.E. & García-García, F.J., 2015. Response of intertidal sandy-beach macrofauna to human trampling: An urban vs. natural beach system approach. *Marine Environmental Research*, **103**, 36-45.

Roberts, C., Smith, C., H., T. & Tyler-Walters, H., 2010. Review of existing approaches to evaluate marine habitat vulnerability to commercial fishing activities. *Report to the Environment Agency from the Marine Life Information Network and ABP Marine*

Environmental Research Ltd. Environment Agency Evidence Report: SC080016/R3., Environment Agency, Peterborough, pp. http://publications.environment-agency.gov.uk/PDF/SCHO1110BTEQ-E-E.pdf

Rostron, D., 1995. The effects of mechanised cockle harvesting on the invertebrate fauna of Llanrhidian sands. In *Burry Inlet and Loughor Estuary Symposium*, pp. 111-117.

Sewell, J. & Sweet, N., 2011. GB Non-native Organism Risk Assessment for *Crepidula fornicata*. www.nonnativespecies.org

Shull, D.H., 1997. Mechanisms of infaunal polychaete dispersal and colonisation in an intertidal sandflat. *Journal of Marine Research*, **55**, 153-179.

Southward, A.J., 1982. An ecologist's view of the implications of the observed physiological and biochemical effects of petroleum compounds on marine organisms and ecosystems. *Philosophical Transactions of the Royal Society of London. B*, **297**, 241-255.

Suchanek, T.H., 1993. Oil impacts on marine invertebrate populations and communities. *American Zoologist*, **33**, 510-523.

Tait, R.V. & Dipper, R.A., 1998. *Elements of Marine Ecology*. Reed Elsevier.

Tenore, K.R., 1977. Growth of *Capitella capitata* cultured on various levels of detritus derived from different sources. *Limnology and Oceanography*, **22** (5), 936-941.

Tenore, K.R. & Chesney, E.J., 1985. The effects of interaction of rate of food supply and population density on the bioenergetics of the opportunistic polychaete, *Capitella capitata* (type 1). *Limnology and Oceanography*, **30** (6), 1188-1195.

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.

Tuck, I.D., Hall, S.J., Robertson, M.R., Armstrong, E. & Basford, D.J., 1998. Effects of physical trawling disturbance in a previously unfished sheltered Scottish sea loch. *Marine Ecology Progress Series*, **162**, 227-242.

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

Van Colen, C., Montserrat, F., Vincx, M., Herman, P.M., Ysebaert, T. & Degraer, S., 2008. Macrobenthic recovery from hypoxia in an estuarine tidal mudflat. *Marine Ecology-Progress Series*, **372**, 31-42.

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.

Vandendriessche, S., Derweduwen, J. & Hostens, K., 2015. Equivocal effects of offshore wind farms in Belgium on soft substrate epibenthos and fish assemblages. *Hydrobiologia*, **756** (1), 19-35.

Ward, T.J. & Young, P.C., 1982. Effects of sediment trace metals and particle size on the community structure of epibenthic seagrass fauna near a lead smelter, South Australia. *Marine Ecology Progress Series*, **9**, 136-146.

Warren, L.M., 1977. The ecology of *Capitella capitata* in British waters. *Journal of the Marine Biological Association of the United Kingdom*, **57**, 151-159.

Watson, G.J., Farrell, P., Stanton, S. & Skidmore, L.C., 2007. Effects of bait collection on *Nereis virens* populations and macrofaunal communities in the Solent, UK. *Journal of the Marine Biological Association of the United Kingdom*, **87** (3), 703-716.

Winter, H., Aarts, G. & Van Keeken, O., 2010. *Residence time and behaviour of sole and cod in the Offshore Wind farm Egmond aan Zee (OWEZ)*. IMARES Wageningen UR.

Wright, S.L., Rowe, D., Thompson, R.C. & Galloway, T.S., 2013. Microplastic ingestion decreases energy reserves in marine worms. *Current Biology*, **23 (**23), R1031-R1033.

Wright, S.L., Thompson, R.C. & Galloway, T.S., 2013b. The physical impacts of microplastics on marine organisms: A review. *Environmental Pollution*, **178**, 483-492.

Wu, B., Qian, P. & Zhang, S., 1988. Morphology, reproduction, ecology and isoenzyme electrophoresis of *Capitella* complex in Qingdao. *Acta Oceanologica Sinica*, **7** (3), 442-458.

Zebe, E. & Schiedek, D., 1996. The lugworm *Arenicola marina*: a model of physiological adaptation to life in intertidal sediments. *Helgoländer Meeresuntersuchungen*, **50**, 37-68.