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Hediste diversicolor and *Streblospio shrubsolii* in littoral sandy mud

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

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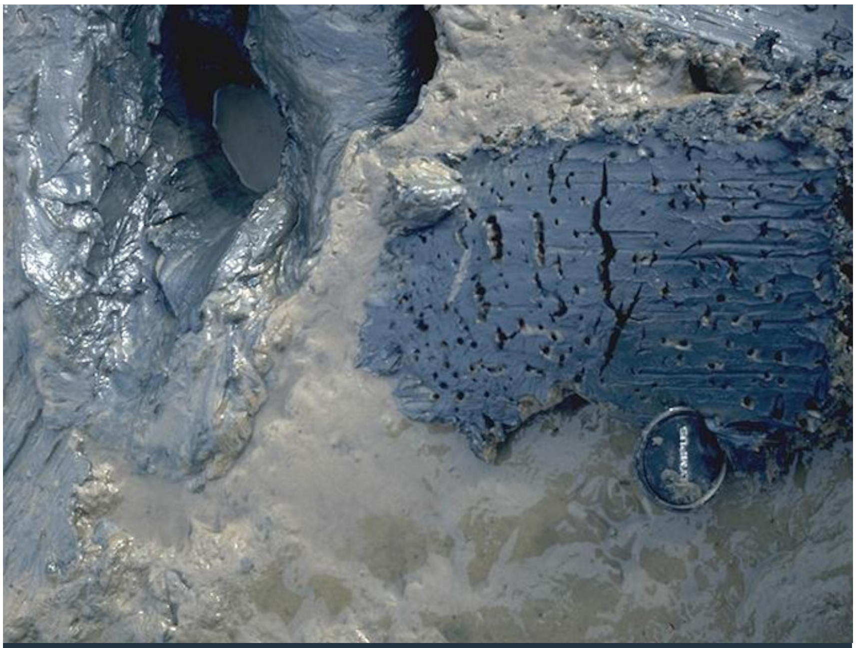

Hediste diversicolor **and** ***Streblospio shrubsolii*** **in littoral sandy mud**
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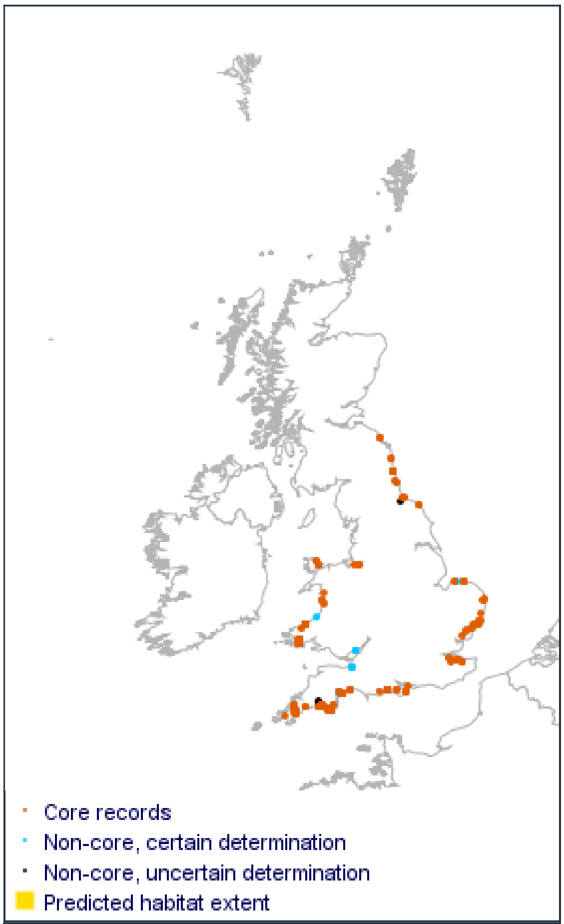

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Biotope distribution data provided by EMODnet Seabed Habitats (www.emodnet-seabedhabitats.eu)

Researched by Matthew Ashley & Georgina Budd **Refereed by** Admin

Summary

UK and Ireland classification

EUNIS 2008	A2.3221	*Hediste diversicolor* and *Streblospio shrubsolii* in littoral sandy mud
JNCC 2015	LS.LMu.UEst.Hed.Str	*Hediste diversicolor* and *Streblospio shrubsolii* in littoral sandy mud
JNCC 2004	LS.LMu.UEst.Hed.Str	*Hediste diversicolor* and *Streblospio shrubsolii* in littoral sandy mud
1997 Biotope	LS.LMU.Mu.HedStr	*Hediste diversicolor* and *Streblospio shrubsolii* in sandy mud or soft mud shores

Description

Mud and sandy mud shores in sheltered marine inlets and estuaries subject to variable or reduced salinity. The biotope (LS.LMu.UEst.Hed.Str) is typically found on the mid and lower shores and is often associated with shallow layers of cobbles and pebbles in the sediment in the upper and mid estuary. The sediment is anoxic close to the surface and remains water saturated during low tide. The infaunal polychaete community is dominated by dense *Hediste diversicolor*, as well as species with a limited salinity range tolerance such as *Streblospio shrubsolii* and *Manayunkia aestuarina*. Oligochaetes, including *Heterochaeta costata* and *Tubificoides benedii* are often abundant, and the amphipod *Corophium volutator* is often common. Hed.Str may occur on the same shores as HedLimScr, HedLim, NhomAph or Hed.Cvol. Higher up on the shore, and/or further towards the head of the estuary, Hed.Ol may occur, changing to Tben at the upper extreme of the estuary. *Ulva* spp. or *Ulva lactuca* may form mats on the surface of the sediment during the summer months, particularly in areas of freshwater influence and/or where there is nutrient enrichment. (Information taken from Connor *et al*., 2004; JNCC, 2015).

Depth range

Mid shore, Lower shore

Additional information

-

Listed By

- none -

Further information sources

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Sensitivity review

Sensitivity characteristics of the habitat and relevant characteristic species

Mud and sandy mud shores in sheltered marine inlets and estuaries subject to variable or reduced salinity. This biotope LS.LMu.UEst.Hed is typically found on the mid and lower shores in the upper and mid estuary. If present on the upper shore, the sediment may become firm and compacted as water drains out, though usually the biotope occurs lower on the shore and the sediment remains water saturated during low tide. An anoxic layer occurs within the upper 5 cm of the sediment. The infauna is dominated by abundant or superabundant ragworms *Hediste diversicolor*. Other species that occur in a significant number of samples include oligochaetes such as *Heterochaeta costata* and *Tubificoides* spp., polychaetes such as *Streblospio shrubsolii* and *Manayunkia aestuarina*, the mud shrimp *Corophium volutator*, and the spire shell *Hydrobia ulvae*. The sensitivity of *Hediste diversicolor* is reviewed in detail, and loss of this species would result loss of the biotope as described in the classification. The sensitivity of oligochaetes (e.g. *Tubificoides spp.*), polychaetes *Streblospio shrubsolii* and *Manayunkia aestuarina and* the mud shrimp *Corophium volutator*, and the spire shell *Hydrobia ulvae* are considered where specific pressures have impacts.

The sediment type in this biotope is the important structural component, providing the complexity required by the associated community. Sediment deposition and, therefore, the spatial extent of the biotope, is dictated by the physiography and underlying geology coupled with the hydrodynamic regime (Elliot *et al*., 1998). There is a traditional view that the distribution of infaunal invertebrates is correlated solely with sediment grain size. In reality, it is likely that a number of additional factors, including organic content, microbial content, food supply and trophic interactions, interact to determine the distribution of the infauna (Snelgrove & Butman, 1994).

This sensitivity review also reflects its three sub-biotopes, which are very simialar in physical charactersitics and species composition, although any differences in potential sensitivity are highlighted where possible. LS.LMu.UEst.Hed.Str, occurs higher up on the shore, and/or further towards the head of the estuary, this mud and sandy mud biotope is typically found on the mid and lower shores and is often associated with shallow layers of cobbles and pebbles in the sediment in the upper and mid estuary. The sediment is anoxic close to the surface and remains water saturated during low tide. The infaunal polychaete community is dominated by dense *Hediste diversicolor*, as well as species with a limited salinity range tolerance such as *Streblospio shrubsolii* and *Manayunkia aestuarina* (Conner *et al.* 2004). LS.LMu.UEst.Hed.Cvol also occurs in sandy mud, which may become firm and compacted if present in the upper shore where there is more time for drainage between high tides. An anoxic layer is usually present within the first 5 cm of the sediment. The infauna is very sparse, usually, only the ragworm *Hediste diversicolor* and the amphipod *Corophium volutator* are present in any abundance. LS.LMu.UEst.Hed.Ol. is typified by a species-poor community, found in mud or slightly sandy mud in low salinity conditions, typically at the head of estuaries. The infauna is dominated by the ragworm *Hediste diversicolor* which is typically superabundant. Oligochaetes, including tubificids and *Heterochaeta costata*, can be abundant, as well as spionids (Conner *et al.* 2004; JNCC, 2015).

Resilience and recovery rates of habitat

When impacted this biotope may recover through the repair of damaged individuals, adult migration by mobile species and recolonization by pelagic larvae. The ability of postlarvae, larger juveniles, and adults of the key characterizing species *Hediste diversicolor* to swim, burrow and be carried by bedload transport can aid the rapid recolonization of disturbed sediments (Shull, 1997).

Davey & George (1986) found evidence that larvae of *Hediste diversicolor* were tidally dispersed within the Tamar Estuary over a distance of 3 km. Such passive dispersal alone suggested that recolonization of disturbed sediments or populations removed from bait digging was likely to occur rapidly, depending upon larvae transport pathways.

Hediste diversicolor is reported to reach maturity between one and three years of age and, like other Nereidae, *Hediste diversicolor* is monotelic, that is, it reproduces only once in its lifetime and then dies (Olive & Garwood, 1981). Populations appear to show local characteristics in terms of spawning periods. Spawning may be limited to a short period in spring or extend over the summer. In the Thames Estuary, Dales (1950) reported specimens growing to maturity within one year, spawning in February, with some individuals surviving up to 18 months. Mettam *et al*. (1982) reported that *Hediste diversicolor* from the Severn Estuary matured rapidly in the spring and spawned at two years old. Olive & Garwood (1981) found that females in the Blyth Estuary, Northumberland, were in their second year before eggs began to appear, so most probably spawned in their third year.

Mature males crawl around outside their burrows in search of a mature female and discharge sperm through the nephridia, directly outside her burrow. Direct contact between the sexes is not a necessity. Sperm is drawn into the burrow by females and fertilized eggs remain inside the burrow protected by the female. Both sexes die shortly after spawning. The trait to lay and protect eggs within a burrow is likely to increase the time populations recover from pressures such as sediment removal, as both adults and eggs will be affected. The pelagic larval dispersal phase is short (Scaps, 2002).

In general, recovery of *Hediste diversicolor* populations from impacts appears to be relatively rapid. Recovery will be enhanced where adult migration (active or passive) can transport adults from adjacent, unimpacted habitats. Where a large area is severely impacted, however, recovery may require longer time-scales. The effects of pipeline construction on benthic invertebrates were investigated using a before/after impact protocol at Clonakilty Bay, West Cork, Ireland. Benthic invertebrates were sampled once before the excavation and at one, two, three and six months after the completion of the work. An impact was obvious in the construction site in that no live invertebrates were found at one month after disturbance, but there followed a gradual recolonization by *Hediste diversicolor*. At six months after the disturbance, there was no significant difference in the mean number of total individuals (of all species) per core sample amongst all study sites but the apparent recovery in the impacted area was due to recovery of *Hediste diversicolor* and *Tubifex* spp. (Lewis *et al*., 2002b).

Bolam *et al*. (2004) experimentally simulated (in the field) the effect of dredged material emplacement (beach recharge) by manipulating defaunated sediments. Macrofaunal sampling was carried out after 1 week and after 1, 3, 6 and 12 months. Recolonization patterns, although gradual, were found to be species-specific with abundances of the polychaete *Hediste diversicolor* and the gastropod *Hydrobia ulvae* recovering to ambient levels within one week.

Corophium volutator lives for a maximum of one year (Hughes, 1988) and females can have 2-4 broods in a lifetime (Conradi & Depledge, 1999). Populations in southerly areas such as the Dovey Estuary, Wales or Starrs Point, Nova Scotia have two reproductive episodes per year. Those populations in colder, more northerly areas such as the Ythan Estuary, Scotland or the Baltic Sea only have one (Wilson & Parker, 1996). On the west coast of Wales, breeding takes places from April to October and mating takes place in the burrow. Adult males crawl over the surface of the moist sediment as the tide recedes in search of burrows occupied by mature females. *Corophium* *volutator* forms an important food source for several species of birds and mobile predators such as fish and crabs (Hughes, 1988; Jensen & Kristensen, 1990; Raffaelli *et al*., 1991; Flach & De Bruin, 1994; Brown *et al*., 1999), so this behaviour makes them vulnerable to predation (Fish & Mills, 1979; Hughes, 1988; Forbes *et al*., 1996). The females can produce 20-52 embryos in each reproductive episode (Fish & Mills 1979; Jensen & Kristensen, 1990). Juveniles are released from the brood chamber after about 14 days, and development is synchronized with spring tides, possibly to aid dispersal. Recruitment occurs within a few centimetres of the parent, although they may disperse later by swimming (Hughes, 1988). In the warmer regions where *Corophium volutator* is found, juveniles can mature in two months (Fish & Mills, 1979) and add their own broods to the population. The juveniles born in May undergo rapid growth and maturation to reproduce from July to September and generate the next overwintering population (Fish & Mills, 1979).

Usually for oligochaetes fertilization is internal and relatively few large eggs are shed directly into a cocoon that is secreted by the worm (Giere & Pfannkuche, 1982). Asexual reproduction is possible in some species by spontaneous fission (Giere & Pfannkuche, 1982). The naid oligochaete *Paranais litoralis* can produce asexually producing clones. The rapid rate of increase (18 times population abundance in 3 months, Gillett *et al*., 2007) allows this species (which is sensitive to high temperatures, hypoxia and is exposed to predation due to shallow burial) to repopulate rapidly when conditions are favourable. However, few Tubificidae and Enchytraeidae produce asexually (Giere & Pfannkuche, 1982). Tubificid populations tend to be large and to be constant throughout the year, although some studies have noticed seasonal variations (Giere & Pfannkuche, 1982). Many species, including *Tubificoides benedii* and *Baltidrilus costata*, have a two-year reproductive cycle and only part of the population reproduces each season (Giere & Pfannkuche, 1982). Populations of *Tubificoides benedii* in the Forth estuary have not demonstrated clear seasonality in recruitment (Bagheri & McLusky, 1982), although mature *Tubificoides benedii* (as *Peloscolex benedeni*) in the Thames Estuary were reported to occur in December with a maximum in late February (Hunter & Arthur, 1978), breeding worms increased from April and maximum cocoon deposition was observed in July (Hunter & Arthur, 1978). Bolam & Whomersley (2003) observed faunal recolonization of fine sediments placed on saltmarsh as a beneficial use and disposal of fine-grained dredged sediments. They found that tubificid oligochaetes began colonizing sediments from the first week following a 'beneficial use' scheme involving the placement of fine-grained dredged material on a salt marsh in south-east England. The abundance of *Tubificoides benedii* recovered slowly in the recharge stations and required 18 months to match reference sites and those in the recharge stations before placement of sediments. The results indicate that some post-juvenile immigration is possible and that an in-situ recovery of abundance is likely to require more than 1 year. Rapid recolonization has also been observed in the tubificid oligochaete *Baltidrilus costata (Tubifex costatus)* that appeared in upper sediment layers in experimentally defaunated patches ($4m^2$) after 3 weeks (Gamenick *et al*., 1996).

Resilience assessment. Resilience is assessed as **'High'** (within 2 years), where resistance is 'High', 'Medium' or 'Low' based on Bolam *et al*. (2004) and Bolam & Whomersley (2003). Resilience is assessed as **'Medium'** (2-10 years) where resistance is 'None' and habitat recovery may also be required. An **exception** is made for permanent or ongoing (long-term) pressures where recovery is not possible as the pressure is irreversible, in which case resilience is assessed as **'Very low'** by default.

NB: The resilience and the ability to recover from human induced pressures is a combination of the environmental conditions of the site, the frequency (repeated disturbances versus a one-off event) and the intensity of the disturbance. Recovery of impacted populations will always be mediated by stochastic events and processes acting over different scales including, but not limited to, local habitat conditions, further impacts and processes such as larval-supply and recruitment between populations. Full recovery is defined as the return to the state of the habitat that existed prior to impact. This does not necessarily mean that every component species has returned to its prior condition, abundance or extent but that the relevant functional components are present and the habitat is structurally and functionally recognizable as the initial habitat of interest. It should be noted that the recovery rates are only indicative of the recovery potential.

Hydrological Pressures

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

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 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). *Hediste diversicolor* and other important characterizing species are adapted to living within the intertidal zone where temperatures fluctuate. Some resistance to temperature fluctuations is achieved by burying within the sediment, which buffers against acute temperature changes over the tidal cycle.

The geographic range of *Hediste diversicolor* (throughout north-west Europe on the Baltic Sea, North Sea and along Atlantic coasts to the Mediterranean) suggests that it is tolerant of a range of temperatures and a temperature increase at benchmark levels is unlikely to adversely effect on UK populations. *Hediste diversicolor* can tolerate temperatures from below zero under Baltic ice to high summer temperatures in Black Sea lagoons (>25°C) (Smith, 1977).

Hediste diversicolor was not strongly affected by heatwaves in an estuary in northwestern Portugal, where temperatures reached 40°C in intertidal pools (higher temperatures than experienced around the UK and Irish coasts) (Dolbeth *et al.*, 2011). Temperature change may adversely affect reproduction. Bartels-Hardege and Zeeck (1990) demonstrated that an increase from 12°C and maintenance of water temperature at 16°C induced reproduction in *Hediste diversicolor* specimens outside the normal period of spawning, and without a drop in temperature to simulate winter conditions the spawning period was prolonged and release of gametes was not synchronized. Poor synchronization of spawning could result in reduced recruitment, as gametes are wasted and mature specimens die shortly after gamete release.

Streblospio shrubsolii have been shown to reproduce in a temperature range of 7.5°C – 30°C with highest reproduction levels occurring between 16°C – 21°C (Levin & Creed, 1986, Da Fonseca-Genevois & Cazaux, 1987, Chu & Levin, 1989, Lardicci *et al.*, 1997). The case studies returned by literature reviews were based on Mediterranean case study sites, limiting confidence for the UK and Irish seas. The timing of reproduction and growth, although occurring throughout the year increased in late spring and early summer, but were strongly reduced during periods of higher temperatures in summer and disappeared or strongly reduced at lower temperatures in winter and (Lardicci *et al.*, 1997). *Streblospio shrubsolii* show timing of growth and reproduction depends on synergistic effects of temperature and photoperiod, suggesting, temperature and photoperiod cues may differ at locations at different latitudes (Chu & Levin, 1989). Both a 5°C increase in temp for one month period or 2°C for one year are within the temperature range reproduction occurs within (7.5°C – 30°C) and within the temperature range where highest reproduction levels occur (16°C – 21°C). Timing and level of reproduction may be affected, however, with some limit on population abundance likely to be caused by a 5°C increase in temp for one month period but this would be unlikely to significantly impact the biotope.

Corophium volutator is widely distributed in the North Atlantic, American and European coasts, from western Norway to the Mediterranean and the Black Sea and Azov Sea (Neal & Avant, 2006). The amphipod is subject to temperatures of 1°C in the winter to 17°C in the summer (Wilson & Parker, 1996) but can resist much higher temperatures (Meadows & Ruagh, 1981).

Tubificoides benedii has increased in abundance in mudflat habitats in Jade Bay, the North Sea between 1930 and 2009 (Schueckel & Kroencke, 2013). Climate warming as well as decreasing nutrient loads and species introductions have occurred in the region since the 1970s, suggesting the species may adapt to temperature increases at benchmark pressures. Bamber & Spencer (1984) observed that *Tubificoide*s were dominant species in an area affected by thermal discharge in the River Medway estuary. Sediments were exposed to the passage of a temperature front of approximately 10°C between heated effluent and estuarine waters during the tidal cycles.

Sensitivity assessment. As characterizing species show limited impacts and potentially show benefits to abundance and recruitment from European case studies (North sea) resistance is assessed as '**High**', resilience is assessed as '**High**', and sensitivity as **'Not Sensitive'.**

Temperature decrease (local)	High Q: High A: High C: High	High Q: High A: High C: High	Not sensitive Q: High A: High C: High

The characterizing species *Hediste diversicolor* is adapted to living within the intertidal zone. Some resistance to temperature fluctuations is achieved by burying within the sediment, which buffers against acute temperature changes over the tidal cycle.

A decrease in temperature has been shown to be indirectly beneficial to *Hediste diversicolor* as case studies report a reduction in numbers of the species' predators. For instance, a severe winter in the Wadden Sea in 1995/1996 saw an increased abundance of *Hediste diversicolor* coinciding with a reduction in the numbers of *Carcinus maenas* and *Crangon crangon* (Armonies *et al*., 2001). A similar increase in abundance was noted in the same area between 1978 and 1987 after a series of cold winters the mean abundance of *Hediste diversicolor* increased from 24/m to 151/m (Beukema, 1990). A similar increase in abundance was noted in the same area between 1978 and 1987 after a series of cold winters (mean *Hediste diversicolor* density increased from 24/m to 151/m respectively) (Beukema, 1990). Decreased temperatures throughout the year may limit reproduction. Bartels-Hardege & Zeeck (1990) induced spawning in the laboratory, in specimens of *Hediste diversicolor* from tidal flats of the Jadebusen (North Sea), outside the normal spawning period of early spring. Temperatures were not lowered to simulate winter conditions but maintained at 16°C. Mature specimens appeared after four weeks and released gametes after a further four weeks according to a semilunar cycle. Reproduction was sustained for a period of four months. Such an extended spawning was witnessed on the Jadebusen following an unusually warm winter. Spawning occurred from February until May and was less synchronized. In contrast, the same population spawned within two months (February - March) following lower winter temperatures in another year. They concluded that not only a threshold temperature was important for synchronized spawning but the timing of the rise in temperature following winter was also a significant factor (Bartels-Hardege & Zeeck, 1990). Hence, a reduced rise in temperature is likely to affect spawning.

Corophium volutator is subject to temperatures of 1°C in the winter to 17°C in the summer (Wilson & Parker, 1996). The population may reduce activity and delay reproduction if the temperature drops below 7°C. Sudden pulses of very cold water can disrupt the circa-tidal rhythms of *Corophium volutator* by resetting the onset of swimming behaviour. For example, a six-hour cold spell would lead to the population trying to swim at low tide and leave them vulnerable to increased predation. However, it took temperatures of 15-20°C below ambient temperature to induce this response (Holmström & Morgan, 1983b). Drolet *et al.* (2013) sampled two intertidal mudflats in the upper Bay of Fundy, Canada, over two consecutive winters (2009–2011), where sediment temperature, 5 cm deep, reached ▯2°C. The authors concluded *Corophium volutator* survived winter through physiological adaptations

Streblospio shrubsolii has been shown to reproduce in a temperature range of 7.5°C to 30°C with highest reproduction levels occurring between 16°C to 21°C (Levin & Creed, 1986, Da Fonseca-Genevois & Cazaux, 1987, Chu & Levin, 1989, Lardicci *et al.*, 1997). Reproductive activity disappeared or strongly reduced at lower temperatures in winter in a Mediterranean case study (Lardicci *et al.*, 1997). Their case study suggests reproduction would be delayed in the UK and Irish populations that experienced both a 5°C decrease in temp for one month period, or 2°C for one year.

Sensitivity assessment. As characterizing species show limited impacts and potentially show benefits to abundance and recruitment from European case studies (e.g. in the North Sea) resistance is assessed as '**High**', resilience is assessed as '**High**', and sensitivity as '**Not Sensitive**'.

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

The salinity ranges in which the variant sub-biotopes are found vary. LS.LMu.UEst.Hed.OL is found in reduced and variable salinity (JNCC, 2015), LS.LMu.UEst.Hed.Str is found in variable salinity (JNCC,2015) and LS.LMu.UEst.Hed.Cvol is found in full reduced and variable salinities. Although salinity is a chemical factor that structures the biological assemblage, other factors such as height on the shore are also likely to be important in determining habitat suitability for the species that characterize the biotopes.

Environmental fluctuations in salinity are only likely to affect the surface of the sediment and not deeper buried organisms since the interstitial or burrow water is less affected. However, under a longer-term or permanent increase in salinity, sediment waters would be expected to also adjust.

Hediste diversicolor, the key characterizing species, occurs across all variant sub-biotopes and as such is resistant to the salinity range, from reduced to full, that the various sub-biotopes occur within. *Hediste diversicolor* is a euryhaline species, able to tolerate a range of salinities from fully marine seawater down to 5 psu or less (Barnes, 1994).

Streblospio shrubsolii occurred in subtidal areas of the Thames estuary as well as intertidal flats, suggesting the species is resistant to higher salinities than the 'variable' levels occurring higher in estuaries (Attrill, 1998). Likewise, *Tubificoides benedii* has been recorded in high abundance in offshore areas of the North Sea (Gray *et al.*, 1990). Although evidence was limited on the response of these species to rapid increases in salinity it is likely they would be resistant to an increase to the fully marine category (30-40‰). Giere & Pfannkuche (1982) identified how species change over a hypothetical salinity gradient with marine stenohaline species present at full salinities replaced by more euryhaline oligochaete species including *Tubificoides benedii* and *Tubificoides pseudogaster*, *Paranais litoralis* and *Baltidrilus costata* (formerly *Heterochaeta costata*). Studies in the Rhine delta have found that *Tubificoides benedii*, is more tolerant of a range of salinities than *Baltidrilus costata* (as *Heterochaeta costata*) which preferred shallow water brackish stations (Verdonschot *et al*., 1982). However, numerous studies suggest that *Baltidrilus costata* tolerates a wide range of salinities from 1‰ to 28‰ (Giere & Pfannkuche, 1982), suggesting that while tolerant of some changes, an increase to full salinity may lead to reductions in abundance of this species.

Hylleberg (1975) also found that under controlled conditions of salinity ranging from 10 to 30□⁄□□and temperatures ranging from 5 to 35° C, *Hydrobia ulvae* has maximal egestion at the combination of high salinity (30□⁄□□) and high temperature (30°C). The species would be likely to show high resistance to an increase in salinity from the reduced and variable conditions the biotope occurs within.

The amphipod *Corophium volutator* occupies a greater variety of habitats and is found in fine sediments of mudflats, salt marsh pools and brackish ditches. It is therefore exposed in its natural range to salinity extremes, from freshwater flushing during periods of high rainfall and increases in salinity during periods of high evaporation. *Corophium volutator* is, therefore, an exceptionally euryhaline species able to tolerate 2-50 psu (McLusky, 1968) but growth is fastest at 15-20 psu (McLusky, 1967; McLusky, 1970 cited in Meadows & Ruagh, 1981). The interstitial salinity is more important for *Corophium volutator* than that of the overlying water. Sustained periods of increased salinity are required to alter that of the interstitial water and there is a lag between salinity changes and the response of *Corophium volutator* (McLusky, 1968). Sudden increases in salinity delay swimming activity (Harris & Morgan, 1984a). *Corophium volutator* will also migrate from areas of unfavourable salinity (McLusky, 1968).

Sensitivity assessment. *Hediste diversicolor* and other characterizing species are likely to tolerate increased salinity levels above the reduced and variable levels encountered in the biotope and variant sub-biotopes. Biotope resistance is, therefore, assessed as '**High**' and resilience as '**High**' (by default) and the biotope is considered to be **'Not sensitive'**.

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

LS.LMu.UEst.Hed.Str is found in variable salinity (JNCC,2015), while LS.LMu.UEst.Hed.OL and LS.LMu.UEst.Hed.Cvol are recorded in reduced and variable salinity (JNCC, 2015). The available evidence (summarised below) suggests that the characterizing species are tolerant of a short-term decrease to reduced or low salinity.

The key characterizing species *Hediste diversicolor* is known to tolerate low salinities below 18-24 psu and it has been shown to replace *Arenicola marina* in areas influenced by freshwater runoff or input (e.g. the head end of estuaries) (Barnes, 1994; Hayward, 1994). Lower salinities (<8 psu) can, however, adversely affect on *Hediste diversicolor* reproduction (Ozoh & Jones, 1990; Smith 1964). Fertilization in *Hediste diversicolor* is adapted to high salinity but not to low salinity below 7.63‰ (Ozoh & Jones, 1990). A decrease in salinity at the benchmark pressure (reduction to <18‰) may negatively impact recruitment and abundance if the dilution is close to that threshold.

Oligochaete dominated biotopes are recorded from a range of salinity regimes from full (LS.LSa.MoSa.Ol; LS.LSa.MoSa.Ol.FS), variable (SS.SMu.SMuVS.CapTubi) reduced (SS.SMu.SMuVS.CapTubi; LS.LMu.UEst.Tben) and low (SS.SMu.SMuVS.LhofTtub) habitats (JNCC, 2015). In very low salinities from <15 to 0 ‰, species such as *Limnodrilus* spp. and *Tubifex tubifex* are found (Giere & Pfannkuche, 1982). A decrease in salinity at the pressure benchmark would probably result in replacement by oligochaete species more tolerant of lower salinities such as *Limnodrilus hoffmeisteri* and *Tubifex tubifex* that characterize the low salinity biotope SS.SMu.SMuVS.LhofTtub. Numerous studies suggest that *Baltidrilus costata* tolerates a wide range of salinities from 1‰ to 28‰ (Giere & Pfannkuche, 1982 and references therein), suggesting that this species is likely to still be present in the biotope.

Conde *et al.* (2013) found that *Streblospio shrubsolii* were a dominant species in low salinity, estuarine conditions (5-9‰) in the Tagus estuary, Portugal. In Ria de Aveiro, western Portugal *Streblospio shrubsolii* were characterizing species of communities in estuarine sample sites further upstream with lower salinity, suggesting a high resistance to a decrease in salinity (Rodrigues *et al.*, 2011). Opportunistic, deposit-feeding polychaetes, such as *Streblospio shrubsolii* and *Manayunkia aestuarina* tolerate stressful conditions, and often out-compete more sensitive species due to tolerance of decreased salinity (or other pressures such as organic enrichment). These species may increase in abundance following a decrease in salinity.

Corophium volutator is an exceptionally euryhaline species able to tolerate 2-50 psu (McLusky, 1968) but growth is fastest at 15-20 psu (McLusky, 1970 cited in Meadows & Ruagh, 1981). *Corophium volutator* requires a salinity greater than 5 psu to moult since osmoregulation is lost during moulting (McLusky, 1967). A salinity of at least 7.5 psu is required for reproduction (McLusky, 1968). Mills & Fish (1980), found that *Corophium volutator* is more tolerant of low salinity (2 to 10‰) than *Corophium arenarium*. Females undergoing a pre-copulatory moult failed to lay eggs below salinities of 3‰ (*Corophium volutator*) and 10‰ (*Corophium arenarium*), but in both species, the lowest salinity at which all females moulted and laid eggs was 20‰. Changes in salinity are very unlikely to cause mortality but may alter population distribution and abundances within the biotope as the species is likely to move to more favourable conditions or to suffer some effects on reproduction with resulting decreases in abundance.

Sensitivity assessment. It is considered likely that a decrease in salinity at the pressure benchmark will lead to some species replacement by polychaetes and oligochaetes more tolerant of low salinity. *Hediste diversicolor* and oligochaetes are likely to remain in low salinity conditions. Therefore LS.LMu.UEst.Hed.Str is likely to be replaced by LS.LMu.UEst.Hed.Ol or LS.LMu.UEst.Hed.Cvol in areas exposed to reduced salinity. Resistance is, therefore, assessed as **'Low'.** Resilience (following the restoration of typical conditions) is probably '**High'** and sensitivity is assessed as **'Low'.** It should be noted that resistance would be lower, and sensitivity greater, where salinity was reduced to a level close to freshwater.

Water flow (tidal current) changes (local)	Medium Q: Medium A: Low C: Low	Medium Q: Medium A: Medium C: Medium	Medium Q: Medium A: Low C: Low

This biotope occurs in wave-sheltered areas but no information on the strength of tidal streams was provided in the JNCC (2015) biotope description. *Hediste diversicolor* characteristically inhabits littoral mudflats predominantly of clay (particles < 4 µm), silt (4-63 µm) and to a lesser extent very fine sand (63-125 µm) (Jones *et al.*, 2000). Littoral mud biotopes and the characterizing species occur in very weak (negligible) to weak < 1 knot (<0.5 m/sec.) currents in other biotopes

(JNCC, 2015).

Experimental increases in near-bed current velocity were achieved over intertidal sandflats by placing flumes on the sediment to accelerate water flows (Zuhlke & Reise, 1994). The increased flow led to the erosion of up to 4 cm depth of surface sediments. No significant effect was observed on the abundance of *Capitella capitata* and numbers of *Tubificoides benedii* and *Tubificoides pseudogaster* were unaffected, as they probably avoided suspension by burrowing deeper into sediments. This was demonstrated by the decreased abundance of oligochaetes in the 0-1 cm depth layer and increased abundance of oligochaetes deeper in sediments (Zuhlke & Reise, 1994). A single storm event had a similar result with decreased abundance of oligochaetes in surficial layers, coupled with an increase in deeper sediments (Zuhlke & Reise, 1994). Although *Tubificoides* spp. can resist short-term disturbances their absence from sediments exposed to higher levels of disturbance indicate that they would be sensitive to longer-term changes in sediment mobility (Zuhlke & Reise, 1994). Birtwell & Arthur (1980) reported seasonal changes in abundance in *Baltidrilus costata* (as *Tubifex costatus*) which they attributed to the erosion of the upper sediment layers caused by high river flows and wave action. It should be noted that the experiment refers to sands, which erode at lower velocities than muds as they are less cohesive.

In the turbid waters of estuaries, where many mud habitats develop, a reduction in water flow is likely to result in a significant increase in siltation increasing the silt and clay content of the substratum. Decreases in water flow with increased siltation of fine particles are considered unlikely to alter the physical characteristics of this habitat type as it is already found in sheltered areas where siltation occurs and where particles are predominantly fine. Reductions in water flow through the presence of trestles (for off-bottom oyster cultivation) arranged in parallel rows in the intertidal area (Goulletquer & Héral, 1997) that reduce the strength of tidal currents (Nugues *et al*., 1996) has been observed to limit the dispersal of pseudofaeces and faeces in the water column and thus increase the natural sedimentation process by several orders of magnitude (Ottman & Sornin, 1985, summarised in Bouchet & Sauriau, 2008). As the characterizing oligochaetes can live relatively deeply buried and in depositional environments with low water flows (based on habitat preferences) and low oxygenation they are considered to be not sensitive to decreases in water flow.

Sensitivity assessment. Where increased or decreased water flows altered the sediment type this could lead to sediment reclassification. Such a change is assessed in the sedimentary change assessment. As muds tend to be cohesive and the surface tends to be smooth reducing turbulent flow, an increase at the pressure benchmark may not lead to increased erosion, biotope resistance is assessed as **'Medium'** as a precautionary assessment, resilience is assessed as **'High'** (following restoration of usual conditions) and sensitivity is assessed as **'Low'**. The biotope is not considered to be sensitive to decreased flows due to its presence in sheltered habitats and the tolerance of *Hediste diversicolor* and*Tubificoides benedii* for low oxygen and sediment deposition.

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

The variant sub-biotopes within LS.LMu.UEst.Hed occur from the upper to the lower shore (JNCC, 2015) except for LS.LMu.UEst.Hed.Str, which is found on the lower and mid-shore (JNCC, 2015).

Hediste diversicolor inhabits a burrow within the sediment which may be up to 0.3 m deep. The species retreats within the burrow during periods of exposure, protecting it from desiccation although increased emergence may cause a decline in the abundance of *Hediste diversicolor* at the upper limits of the intertidal zone, as they may become stressed by desiccation if the substrata begin to dry and are prone to more extremes of temperature. *Hediste diversicolor* is sufficiently mobile to gradually retreat to damper substrata. Gogina *et al.* (2010a) analysed patterns of benthic community distribution related to selected environmental parameters, including depth, in the western Baltic Sea with depths ranging from 0 m to 31 m. *Hediste diversicolor* displayed a preference for low-saline regions shallower than 18 m. Increased depth had the largest negative effect of all factors influencing distribution and abundance decreased with greater dept) Gogina *et al.* (2010a).

Tubificoides benedii is capable of burrowing to depths of 10 cm, shows resistance to hypoxia and is often typified as an 'opportunist' that is adapted to the rapid environmental fluctuations and harsh conditions in estuaries (Gogina *et al.*, 2010a). Highest abundances were predicted by Gogina *et al.* (2010a) to be related to depth with an optimum of 10 m to 20 m. The evidence suggests that abundance may be limited by a decrease in high water level or a change in time (increase) where substratum is not covered by the sea. An increase in the time the biotope is covered by the sea is likely to result in increased abundance of *Tubificoides benedii.*

Sensitivity assessment. The biotope and characterizing species are found at a range of shore heights and are considered relatively resistant to changes in emergence that do not alter the extent of the intertidal. An increase in emergence is likely to decrease the upper shore extent of *Hediste diversicolor* dominated biotopes, especially LS.LMu.UEst.Hed.Str, at the landward extent of the intertidal as desiccation increases. A decrease in emergence under the benchmark pressure is likely to extend the upper extent of the biotope as the species recolonize or migrate to favourable conditions. The resistance of the LS.LMu.UEst.Hed.Str biotope is, therefore, assessed as **'Medium',** recoverability is assessed as '**High**' (by default) and the biotope is considered to be **'Low'.**

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

This biotope occurs in wave sheltered areas, where estimated wave categories range from sheltered, very sheltered and extremely sheltered (JNCC, 2015). The key characterizing species *Hediste diversicolor* is infaunal, inhabiting a burrow in which it seeks refuge from predators and may partially emerge to seek and capture food. An alteration of factors within the environment that increases wave exposure could cause erosion of the substrata and consequently, loss of habitat.

Sensitivity assessment. Resistance to a change in nearshore significant wave height >3% but <5% of the characterizing species *Hediste diversicolor* is '**High**', given that the biotope occurs in very sheltered locations and an increase in nearshore significant wave height of >3% but <5% would continue to result in sheltered conditions which are within the species' tolerance limits. At the highest benchmark pressure (5% increase) the species exhibit '**High**' resistance through their traits to live relatively deep in the sediment. Resilience (recoverability) is also '**High**' giving a biotope sensitivity of '**Not Sensitive**'. Due to limited evidence, confidence in this assessment is 'Low'.

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 any evidence is presented where available. The following review discusses impacts at higher concentrations than the pressure benchmark.

Contamination at levels greater than the pressure benchmark may adversely affect the biotope. Bryan (1984) reported that short-term toxicity in polychaetes was highest to Hg, Cu and Ag, declined with Al, Cr, Zn and Pb with Cd, Ni, Co and Se being the least toxic. It was recorded that polychaetes have a range of tolerances to heavy metals levels of Cu, Zn, As and Sn being in the order of 1500-3500 µg/g.

Hediste diversicolor is known to be relatively tolerant of chemicals in estuarine environments, having a role on metal speciation through bioturbation, particle mixing, and irrigation (Dedeh *et al.* 2014; Berthet *et al*. 2003; Mouneyrac *et al.* 2003). *Hediste diversicolor* has been found successfully living in estuarine sediments contaminated with copper ranging from 20 µm Cu/g in low copper areas to >4000 µm Cu/g where mining pollution is encountered e.g. Restronguet Creek, Fal Estuary, Cornwall (Bryan & Hummerstone, 1971). Attempts to change the tolerance of different populations of *Hediste diversicolor* to different sediment concentrations of copper have shown that it is not readily achieved suggesting that increased tolerance to copper has a genetic basis (Bryan & Hummerstone, 1971; Bryan & Gibbs, 1983).

Hediste diversicolor was not affected by lower environmentally relevant concentrations but displayed DNA alterations and significant increase in Cd and Hg concentrations when exposed to metal mixtures at strongest environmentally relevant concentrations out of two experimental mixtures (Cd 17 Ug/l, Hg 1.1 Ug/l, Pb 55 Ug/l) (Dedeh *et al*., 2014). DNA damage was also observed in *Hediste diversicolor* exposed to soluble Ag in the form of nanoparticles (used in a range of products and processes for their antibacterial properties, electric and thermal conductivity) (Buffet *et al*., 2014) and burrowing decreased when exposed to soluble Cu (Buffet *et al*., 2014).

In *Hediste diversicolor*, the acute toxicity is dependent on the rate of uptake of the metal since this determines the speed with which the lethal dose is built up. The rate of intake is important because this determines whether the organism's detoxification mechanisms can regulate internal concentrations. The resistance of *Hediste diversicolor* is thought to be dependent on a complexing system which detoxifies the metal and stores it in the epidermis and nephridia (Bryan & Hummerstone, 1971; McLusky *et al*., 1986).

Crompton (1997) reviewed the toxic effect concentrations of metals to marine invertebrates. Annelid species, such as *Hediste diversicolor* were found to be at risk if metals exceeded the following concentrations during 4-14 days of exposure: >0.1 mg Hg l-1, > 0.01 mg Cu l-1, > 1 mg Cd l-1, >1 mg Zn l-1,>0.1 mg Pb l-1, >1 mg Cr L-1, >1 mg As l-1 and >10 mg Ni l-1.

In general, for estuarine animals, heavy metal toxicity increases as salinity decreases and temperature increases (McLusky *et al*., 1986). For example, Fernandez & Jones (1990) calculated 96 hour LC50 Zinc values for *Hediste diversicolor* at four salinities 5, 10, 17.5 and 30 psu at 12°C. The 96 hour LC50 at 17.5 psu and 12°C was 38 mg Zn l-1, while at 5 and 10 psu it was 7 and 19 mg Zn l-1 respectively. Toxicity decreased with increasing salinity. When salinity remained constant at 17.5 psu, but temperature varied, the following 96 hour LC 50 values for Zinc were recorded: 40 mg Zn l-1 at 6°C, 32 mg Zn l-1 at 12°C and 9.1mg Zn l-1 at 20°C. Toxicity increased with increasing temperature. Accumulation of zinc was also greater at the lowest salinities and when the temperature was highest at 20°C. In a parallel experiment, the presence of sediment was found to reduce toxicity and body accumulation of zinc in *Hediste diversicolor*.

Recovery of this species would be influenced by the length of time it would take for the potential habitat to return to a suitable state (e.g. factors such as the decline of bioavailable metals within the marine environment), recolonization by adult and juvenile specimens from adjacent habitats, and the establishment of a breeding population. Since juveniles remain in the infauna throughout their development selection for metal tolerance can be expected to be operative from an early stage (Bryan & Gibbs, 1983).

Corophium volutator is highly intolerant of metal pollution at levels often found in estuaries from industrial outfalls and contaminated sewage. A concentration 38 mg Cu/l was needed to kill 50% of *Corophium volutator* in 96-hour exposures (Bat *et al.*, 1998). Other metals are far more toxic to *Corophium volutator*, e.g. zinc is toxic over 1 mg/l and toxicity to metals increases with increasing temperature and salinity (Bryant *et al.*, 1985b). Mortality of 50% is caused by 14 mg/l (Bat *et al.*, 1998). Although exposure to zinc may not be lethal, it may affect the perpetuation of a population by reducing growth and reproductive fitness. Mercury was found to be very toxic to *Corophium volutator*, e.g. concentrations as low as 0.1 mg/l caused 50% mortality in 12 days. Other metals known to be toxic include cadmium, which causes 50% mortality at 12 mg/l (Bat *et al.*, 1998), and arsenic, nickel and chromium which are all toxic over 2 mg/l (Bryant *et al.*, 1984; Bryant *et al.*, 1985a, 1985b).

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 any evidence is presented where available.

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. Hailey (1995) cited substantial kills of *Nereis*, *Cerastoderma*, *Macoma*, *Arenicola* and *Hydrobia* as a result of the Sivand oil spill in the Humber estuary in 1983.

The 1969 West Falmouth (*America*) spill of Grade 2 diesel fuel documents the effects of hydrocarbons in a sheltered habitat (Suchanek, 1993). The entire benthic fauna including *Hediste diversicolor* was eradicated immediately following the spill and remobilization of oil that continued for a period >1 year after the spill, contributed to much greater impact upon the habitat than that caused by the initial spill. Effects are likely to be prolonged as hydrocarbons incorporated within the sediment by bioturbation will remain for a long time owing to slow degradation under anoxic conditions. Oil covering the surface and within the sediment will prevent oxygen transport to the infauna and promote anoxia as the infauna utilize oxygen during respiration. Although *Hediste diversicolor* is tolerant of hypoxia and periods of anoxia, a prolonged absence of oxygen will result in the death of it and other infauna. McLusky (1982) found that petrochemical effluents released from a point source to an estuarine intertidal mudflat caused severe pollution in the immediate vicinity. Beyond 500 m distance, the effluent contributed to an enrichment of the fauna in terms of abundance and biomass similar to that reported by Pearson & Rosenberg (1978) for organic pollution, and *Hediste diversicolor* was found amongst an impoverished fauna at 250 m from the discharge.

Tubificoides benedii are often abundant in polluted coastal muds (Giere & Rhode, 1987) and *Capitella capitata* increase in abundance in highly polluted sites compared to sites at a distance from contamination (oil drilling waste) (Gray *et al.*, 1990), suggesting these species would increase in abundance under this pressure.

Light fractions (C10 - C19) of oils are much more toxic to *Corophium volutator* than heavier fractions (C19 - C40). In exposures of up to 14 days, light fraction concentrations of 0.1 g/kg sediment caused high mortality. It took 9 g/kg sediment to achieve similar mortalities with the heavy fraction (Brils *et al*., 2002). In the Forth Estuary, *Corophium volutator* was excluded for several hundred metres around the outfalls from hydrocarbon processing plants. Roddie *et al.* (1994) found high levels of mortality of *Corophium* at sites contaminated with crude oil.

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 any evidence is presented where available.

Hediste diversicolor shows sensitivity to pesticides, particularly those used in association with control of sea lice and mosquito larvae in salmon farms. Reports of the effects of synthetic chemicals on *Hediste diversicolor* illustrate that the intolerance of the species is highly dependent upon the molecular structure of the chemical, which determines the chemicals properties and use. For example:

Collier & Pinn (1998) observed significant differences in both the abundance and biomass of a benthic community from the Ythan Estuary, Scotland, experiencing contamination by Ivermectin. Ivermectin is the 22,23-dihydro derivative of avermectin ß which is highly efficient in the treatment of sea lice. *Hediste diversicolor* was the most intolerant species to Ivermectin in the benthic community studied. A rapid decline in both abundance and total biomass of *Hediste diversicolor* occurred within 7 days and with increasing concentration. An Ivermectin concentration of 8.0 mg m▯ caused 100% mortality within 14 days. Davies *et al*., (1998) modelled factors influencing the concentration of Ivermectin reaching the seabed which ranged from 2.2 to 6.6 mg m▯. The upper limit of this range was only slightly less than the concentrations found to be toxic by Collier & Pinn (1998) and Black *et al*. (1997). Davies *et al*. (1998) concluded that there was a significant risk to benthic organisms within a radius of 50 m of salmon farms utilizing Ivermectin and that Ivermectin could accumulate (half-life of Ivermectin in marine sediments > 100 days) within the sediment beyond a single treatment and reach toxic levels.

In contrast, Craig & Caunter (1990) examined the effects of the organosilicon compound, Polydimethylsiloxane (PDMS) in sediment on *Hediste diversicolor*. PDMS fluids are less dense than water and insoluble and form a discrete layer on the surface of the water. In an intertidal environment, PDMS fluids are deposited upon the sediment surface at low tide and into contact with *Hediste diversicolor*. In laboratory tests, exposure to 10,000 mg PDMS per kg of sediment caused no deaths over 96 hours, and exposure to 1,000 mg PDMS per kg of sediment caused no deaths of *Hediste diversicolor* after 28 days.

Corophium volutator is paralysed by pyrethrum based insecticide sprayed onto the surface of the mud (Gerdol & Hughes, 1993) and pyrethrum would probably cause significant mortalities if it found its way into estuaries from agricultural runoff. Nonylphenol is an anthropogenic pollutant that regularly occurs in water bodies, it is an oestrogen mimic that is produced during the sewage treatment of non-ionic surfactants and can affect *Corophium volutator* (Brown *et al*., 1999). Nonylphenol is a hydrophobic molecule and often becomes attached to sediment in water bodies. This will make nonylphenol available for ingestion by *Corophium volutator* in estuaries where much of the riverine water-borne sediment flocculates and precipitates out of suspension to form mudflats. Nonylphenol is not lethal to *Corophium volutator* but does reduce growth and has the effect of causing the secondary antennae of males to become enlarged which can make the amphipods more vulnerable to predators (Brown *et al*., 1999). *Corophium volutator* is killed by 1% ethanol if exposed for 24 hours or more but can withstand higher concentrations in short pulses. Such short pulses, however, have the effect of rephasing the diel rhythm and will delay the timing of swimming activity for the duration of the ethanol pulse (Harris & Morgan, 1984b).

Radionuclide contamination	No evidence (NEv) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	No evidence (NEv) Q: NR A: NR C: NR

No evidence'. Insufficient evidence was found on the effects of radionuclide contamination to assess this pressure.

Beasley & Fowler (1976) and Germain *et al*. (1984) examined the accumulation and transfers of radionuclides in *Hediste diversicolor* from sediments contaminated with americium and plutonium derived from nuclear weapons testing and the release of liquid effluent from a nuclear processing plant. Both concluded that the uptake of radionuclides by *Hediste diversicolor* was small. Beasley & Fowler (1976) found that *Hediste diversicolor* accumulated only 0.05% of the concentration of radionuclides found in the sediment. Both also considered that the predominant contamination pathway for *Hediste diversicolor* was from the interstitial water. However, there is insufficient information available on the biological effects of radionuclides to comment further upon the intolerance of this species to radionuclide contamination.

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**

Some, all be it limited evidence was returned by searches on activated carbon (AC). AC is utilised in some instances to effectively remove organic substances from aquatic and sediment matrices. Lillicrap *et al.* (2015) demonstrate that AC may have physical effects on benthic dwelling organisms at environmentally relevant concentrations at remediated sites.

De-oxygenation	High Q: High A: Medium C: Medium	High Q: High A: High C: High	Not sensitive Q: High A: Medium C: Medium

The littoral muds and muddy sands which *Hediste diversicolor* and the other characterizing species inhabit, tend to have lower oxygen levels than other sediments. *Hediste diversicolor* is resistant to moderate hypoxia (Diaz & Rosenberg, 1995). The successful survival of this species under prolonged hypoxia was confirmed by the resistance experiments of Vismann (1990), which resulted in a mortality of only 15% during a 22-day exposure of *Hediste diversicolor* at 10% oxygen (ca. 2.8 mg O_2 per litre). *Hediste diversicolor* is active at the sediment/water interface where hydrogen sulphide concentrations increase during periods of hypoxia. Vismann (1990), also demonstrated that the high tolerance of *Hediste diversicolor* to hypoxia in the presence of sulphide is enabled by elevated sulphide oxidation activity in the blood. *Hediste diversicolor* may also exhibit a behavioural response to hypoxia by leaving the sediment (Vismann, 1990) which is enhanced in the presence of sulphide. After 10 days of hypoxia (10% oxygen saturation) with sulphide (172-187 μmM), only 35% of *Hediste diversicolor* had left the sediment compared to 100% of *Nereis virens*. Laboratory experiments in the absence of sediments found that *Hediste diversicolor* could survive hypoxia for more than 5 days and that it had a higher tolerance to hypoxia than *Nereis* *virens*, *Nereis succinea* and *Nereis pelagica* (Theede, 1973; Dries & Theede, 1974; Theede *et al*., 1973). Juvenile *Hediste diversicolor* survived hypoxic conditions for 4d in laboratory conditions and combined hypoxia and increased sulphide for 3 days (Gamenick *et al*., 1996). Postlarval *Hediste diversicolor was* the only life stage to show less tolerance to hypoxia, surviving for only 14h (Gamenick *et al*.,1996).

Oligochaete species vary in their tolerance of hypoxia and associated high sulphide levels. Most enchytraeids and naidids are sensitive to hydrogen sulphide and hypoxia while tubificids are often more resistant (Giere, 2006). *Tubificoides benedii* has a high capacity to tolerate anoxic conditions, its extreme oxygen tolerance is based on an unusually low respiration rate (Giere *et al*., 1999). Respiration rates of *Tubificoides benedii* measured at various oxygen concentrations showed that aerobic respiration is maintained even at very low oxygen concentrations (Giere *et al*., 1999). Birtwell & Arthur (1980) showed that *Tubificoides benedii* could tolerate anoxia in the Thames Estuary (LT_{50} = 58.8 hours at 20°C, 26.6 hours at 25°C and 17.8 hours at 30°C in experiments with worms acclimated to 20°C). Tolerance experiments by Gamenick *et al.* (1996) found that *Baltidrilus costata* (as *Heterochaeta costata)* was not affected by hypoxic conditions for at least 3 days but the addition of sulphide 91.96 mmol/litre) caused mortality after 1 day (Gamenick *et al*., 1996)

In a series of experiments Gamenick *et al.* (1996) suggested that *Corophium volutator* is highly sensitive to hypoxia and suffers 50% mortality after just 4 hours in hypoxic conditions, or within 2 hours if there is a rapid build-up of sulphide (Gamenick *et al*., 1996). These results are largely in concordance with other work by Gamble (1970) who found that survival rates were temperature-dependent with individuals surviving longer at lower temperatures. The level of oxygen was not assessed by Gamenick *et al.* (1996) and the description of the experimental set-up suggests that anoxic test conditions were used rather than hypoxic. Gamble (1970) found that at 5°C most individuals were inactive after 30 minutes exposure to anaerobic seawater and that mortality occurred later, the inactivity may have allowed the species to survive longer (Gamble, 1970).

Sensitivity assessment. Resistance to exposure to dissolved oxygen concentration of less than or equal to 2mg/l for 1 week is assessed as **'High'** given the tolerance of the characterizing species *Hediste diversicolor* and oligochaetes, *Corophium volutator* may be more sensitive, however, as the biotope is intertidal the biotope will be oxygenated when immersed reducing effects and *Corophium volutator* may migrate landwards to reduce exposure. Recoverability is assessed a **'High**' (by default) and the biotope is assessed as '**Not sensitive**' to this pressure at the benchmark level.

Nutrient enrichment	High Q: High A: Medium C: Medium	High Q: High A: High C: High	Not sensitive Q: High A: Medium C: Medium

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). Primary production in the biotope will be limited to microalgae at the sediment surface, rather than macroalgae. Changes in primary production as a result of changes in nutrient enrichment are, therefore, not considered likely to directly alter the biotope.

Aberson *et al.* (2016) found nutrient enrichment promotes surface deposit-feeding in *Hediste diversicolor*, over suspension-feeding and predation. At sewage-polluted sites in three estuaries in south-east England *Hediste diversicolor* mainly consumed microphytobenthos, sediment organic matter and filamentous macroalgae *Ulva* spp. At cleaner sites *Hediste diversicolor* relied more on suspension-feeding and consumption of *Spartina anglica* (Aberson *et al*., 2016). Whilst suggesting adaptability to nutrient enrichment this behaviour will increase predation risk.

Nutrient enrichment favours the growth of opportunistic green macro-algae blooms which can cause declines in some species and increases in others (Raffaelli, 2000). Evidence (Beukema, 1989; Reise *et al*., 1989; Jensen, 1992) suggested a doubling in the abundance of *Hediste diversicolor* in the Dutch Wadden Sea, accompanied by a more frequent occurrence of algal blooms that were attributed to marine eutrophication. Algae may be utilized by *Hediste diversicolor* in its omnivorous diet, so some effects of nutrient enrichment may be beneficial to this species. However, evidence for the effects of algal blooms stimulated by nutrient enrichment on *Hediste diversicolor* is not consistent. Raffaelli (1999) examined a 30-year database to examine the effect of nutrient enrichment on an estuarine food web in Aberdeenshire, Scotland. This study displayed impacts to species characterizing the biotope from the development of algal mats, the density and distribution of which was related to nutrient. In areas where algal biomass was greatest, reduced invertebrate densities were recorded. The mud shrimp *Corophium volutator* showed the greatest decrease in density. Densities of *Corophium volutator*, *Limecola balthica* and *Hediste diversicolor* were lower in 1990 compared to 1964 at sites where macro-algal mats increased over the same period. Conversely, densities were on average higher in the upper reaches where macroalgal mats were generally absent before 1990 (Raffaelli, 1999). *Capitella capitata* and *Pygospio elegans* abundance were greater in areas that received the greatest nutrient enrichment (Raffaelli, 1999). Long-term nutrient enrichment may, therefore, alter the biotope if high biomass of algal mats persists.

Sensitivity assessment. The benchmark is relatively protective and is not set at a level that would allow blooms of green algae on the sediment, based on this consideration and based on the lack of primary producers structuring the biotope, resistance is assessed as **'High**' and resilience as **'High'** (by default) so that the biotope is assessed as **'Not sensitive'.**

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

All species apart from two that are abundant in the biotope are classed in a Marine Biotic Index as being indifferent to, tolerating or proliferating under organic enrichment conditions (Borja *et al*., 2000). Only *Limecola balthica* and *Manayunkia aestuarina* are recorded as being sensitive to organic enrichment.

However, case studies demonstrate the resilience of *Limecola balthica* populations to enrichment. *Limecola balthica* (as *Macoma balthica*) have been shown experimentally to be able to resist periods of up to 9 weeks under algal clover, their long siphon allowing them to reach oxygenated water although other bivalves decreased in abundance (Thiel *et al*., 1998). Organic enrichment from waste-water discharge in the Dutch Wadden Sea resulted in positive effects on *Limecola balthica* abundance, biomass, shell growth and production. These effects were concluded to be due to increased food supply (Madsen & Jensen, 1987). *Tubificoides benedii* and other oligochaetes are very tolerant of high levels of organic enrichment and often dominate sediments where sewage has been discharged or other forms of organic enrichment have occurred (Pearson & Rosenberg, 1978; Gray, 1971; McLusky *et al*., 1980).

Sensitivity assessment. At the benchmark levels, a resistance of '**High**' as the main characterizing species *Hediste diversicolor* is tolerant of organic enrichment and 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 habitats and benthic species are considered to have a resistance of '**None**' to this pressure and to be unable to recover from a permanent loss of habitat (resilience is '**Very Low'**).

Sensitivity assessment. 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)	Low Q: High A: High C: High	Very Low Q: High A: High C: High	High Q: High A: High C: High

This biotope is only found in sediment, in particular, sandy mud or mud and the burrowing organisms, *Hediste diversicolor*, would not be able to survive if the substratum type was changed to either a soft rock or hard artificial type. Consequently, the biotope would be lost altogether if such a change occurred.

Sensitivity assessment. Biotope resistance is assessed as '**None',** resilience is '**Very low'** (as the change at the pressure benchmark is permanent) and biotope sensitivity is '**High**'.

Physical change (to another sediment type)	Low Q: High A: Medium C: NR	Very Low Q: High A: High C: High	High Q: High A: Medium 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 biotope occurs in sandy mud and mud and the variant sub-biotopes show some variation in species communities and sediment type (JNCC, 2015) so that some changes in the proportion of finer or coarser sediments may lead to biotope reversions between the sub-biotopes. There may also be broad areas of transition between the biotope LS.LMu.UEst.Hed and between the sub-biotopes of HedMx (JNCC, 2015). The boundaries may be very indistinct, with the HedMx groups present in patches of gravelly mud on areas of mudflat where the main biotopes are their corresponding mud or sandy mud biotopes (JNCC, 2015). Changes in sediment type may, therefore, be part of usual temporal variation within this biotope, so that the assessment (based on no recovery) is precautionary, the confidence levels have been set as 'Low' to reflect this.

Sensitivity assessment. An increase in sand or gravel content is likely to lead to a change to comparable mixed sediment biotopes and biotope reclassification. Resistance to a change in one Folk class is assessed as **'Low'** as biotopes are likely to change to corresponding mixed sediment biotope, while increased gravel content is likely to lead to reduced abundance of characterizing species. Resilience is assessed as **'Very Low**' as a change at the benchmark is permanent. The sensitivity of the biotope is, therefore, considered to be '**High**'.

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

The substratum of this biotope consists of sandy mud or mud (JNCC, 2015). The characterizing infaunal species, including *Hediste diversicolor* and oligochaetes, burrow into the sediment, to depths not exceeding 30 cm. The process of extraction is considered to remove all biological components of the biotope group in the impact footprint and the sediment habitat.

Sensitivity assessment. Resistance to the extraction of the substratum to 30 cm across the entire biotope is assessed as **'None'** based on expert judgment but supported by the literature relating to the position of these species on or within the seabed and literature on impacts of dredging and bait digging activities (see penetration and disturbance pressure). At the pressure benchmark, the exposed sediments are considered to be suitable for recolonization almost immediately after extraction. Recovery will be mediated by the scale of the disturbance and the suitability of the sedimentary habitat, biotope resilience is assessed as **'High'** (based on recolonization by adults and pelagic larvae) and biotope sensitivity is assessed as '**Medium'.**

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

Muddy sand sediments, in general, tend to be cohesive although high levels of water content will reduce this and destabilise sediments. Sediment cohesion provides some sediment stabilisation to resist erosion following surface disturbance. The characterizing species associated with this biotope are infaunal and hence have some protection against surface disturbance. The snail *Hydroia ulvae* is present on the surface and abrasion may result in burial or damage to this species. Surface compaction can collapse burrows and reduce the pore space between particles, decreasing penetrability and reducing stability and oxygen content (Sheehan, 2007). Trampling (3 times a week for 1 month) associated with bait digging reduced the abundance and diversity of infauna (Sheehan, 2007; intertidal muds and sands).

The burrowing life habits of *Corophium volutator* are likely to provide some protection from abrasion at the surface only. However, any abrasion or physical disturbance is likely to reduce the density of *Corophium* spp. by emigration and increased mortality. For example, the sediment turnover caused by cockles and lugworms disturbs the burrows of *Corophium volutator* and caused a significant negative effect on *Corophium volutator* density as a result of an increased rate of swimming making the amphipod more vulnerable to predation (Flach & De Bruin, 1993, 1994). *Corophium arenarium* is also sensitive to sediment disturbance from bioturbating species (Flach, 1993).

Sensitivity assessment. Resistance is assessed as '**Medium',** as abrasion is unlikely to affect high numbers of infaunal burrowing species such as the key characterizing species *Hediste diversicolor* and the oligochaetes but *Hydrobia ulvae*, may be reduced in abundance. Resilience is assessed as '**High'** and biotope sensitivity is assessed as **'Low'.**

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

As the characterizing species are burrowing species, the impact from damage to the sub-surface sea bed would be greater than damage to the sea bed surface only (see abrasion pressure). A number of studies have assessed the impacts of activities resulting in penetration and disturbance of sediments on the characterizing species in similar habitats. The characterizing species have some protective traits such as infaunal life habit, with deeper burrowing species less exposed.

The effects of pipeline construction on benthic invertebrates were also investigated using a 'Before/After' impact protocol at Clonakilty Bay, West Cork, Ireland. Benthic invertebrates were sampled once before the excavation and at one, two, three and six months after the completion of the work. Invertebrate samples were dominated *by Hediste diversicolor*, *Scrobicularia plana* and *Tubifex* spp. An impact was obvious in the construction site in that no live invertebrates were found at one month after disturbance, but there followed a gradual recolonization by *Hediste diversicolor*. At six months after the disturbance, there was no significant difference in the mean number of total individuals (of all species) per core sample amongst all study sites, but the apparent recovery in the impacted area was due to two taxa only, *Hediste diversicolor* and *Tubifex* spp. (Lewis *et al*., 2002).

In the Columbia River, no significant difference was found in *Corophium volutator* densities before and after dredging a channel and no difference between the dredged site and a control site (McCabe *et al*., 1998). Presumably, the dredging did cause mortality of *Corophium volutator* but recolonization was so rapid that no difference was found. The extraction of cockles by sediment raking and mechanical disturbance and digging for lugworms for bait is likely to cause significant mortality of *Corophium volutator*. Bait digging was found to reduce *Corophium volutator* densities by 39%. Juveniles were most affected suffering a 55% reduction in dug areas (Shepherd & Boates, 1999).

Sensitivity assessment. The resistance of the biotope is assessed as **'Low'**, although the significance of the impact for the bed will depend on the spatial scale of the pressure footprint. 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

Changes in light penetration or attenuation associated with this pressure are not relevant to *Hediste diversicolor* and *Limecola balthica* biotopes. As the species live in the sediment they are also likely to be adapted to increased suspended sediment (and turbidity). However, alterations in the availability of food or the energetic costs in obtaining food or changes in scour could either increase or decrease habitat suitability for *Hediste diversicolor*, *Limecola balthica* as characterizing species and for other abundant species such as the oligochaetes *Tubificoides benedii* and *Tubificoides pseudogaster*.

Hediste diversicolor characteristically inhabits estuaries where turbidity is typically higher than other coastal waters. Changes in the turbidity may influence the abundance of phytoplankton available as a food source that may be attained through filter feeding. *Hediste diversicolor* utilizes various other feeding mechanisms and, at the benchmark level, the likely effects of a change in one rank on the WFD scale are limited.

Sensitivity assessment. Resistance is '**High**' as no significant negative effects are identified and potential benefits from increased food resources may occur. Resilience is also '**High**' as no recovery is required under the likely impacts. The sensitivity of the biotope is, therefore, assessed as '**Not Sensitive'**. The sensitivity assessment relies on expert judgement, utilising evidence of species traits and distribution and, therefore, confidence has been assessed as 'Low'.

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

The biotope is located mainly in mid and lower shore sandy mud or mud in lower estuaries, sheltered bays and marine inlets (Connor *et al.*, 2004). These locations would be likely to experience some redistribution of fine material during tidal cycles. Although the biotope occurs in sheltered locations some mixing from wave action may also be expected. The characterizing species *Hediste diversicolor* live in the sediment, to depths to 15cm and 5-6 cm respectively (Esselink & Zwarts, 1989; Stekoll *et al.*, 1980, Volkenborn & Reise, 2006) and would be expected to be well adapted to these conditions.

Long-term deposition of fine material (e.g. continuous deposition) would be expected to lead to higher densities of macrobenthic organisms. For example, in the North Sea (Belgium) deposition of fine particle sediment, disturbed by scour around the base of a wind farm tower led to higher macrobenthic densities and created a shift in macrobenthic communities around the wind farm tower (influenced by the direction fine material had settled) (Coates *et al.*, 2014). The degree to which the characterizing species can resist this pressure depends primarily on species mobility, ability to survive within sediment without contact with the surface and ability to escape from the over-burden. Factors that affect the ability to regain the surface include grain size (Maurer *et al.*, 1986), temperature and water content (Chandrasekara & Frid, 1998).

Mobile polychaetes have been demonstrated to burrow through thick layers of deposits. Powilleit *et al.*, (2009) studied the response of the polychaete *Nephtys hombergii* to smothering. This species successfully migrated to the surface of 32-41 cm deposited sediment layer of till or sand/till mixture and restored contact with the overlying water. While crawling upward to the new sediment surfaces burrowing velocities of up to 20 cm/day were recorded for *Nephtys hombergii*. Similarly, Bijkerk (1988, results cited from Essink 1999) indicated that the maximal overburden through which species could migrate was 60 cm through mud for *Nephtys* and 90 cm through sand. No further information was available on the rates of survivorship or the time taken to reach the surface.

Tubificoides spp. and other oligochaetes live relatively deeply buried and can tolerate periods of low oxygen that may occur following the deposition of a fine layer of sediment. *Tubificoides* spp. showed some recovery through vertical migration following the placement of a sediment overburden 6cm thick on top of sediments (Bolam, 2011). Whomersley *et al.*, (2010) experimentally buried plots on intertidal mudflats at two sites (Creeksea- Crouch Estuary, England and Blackness- lower Forth Estuary, Scotland), where *Tubificoides benedii* were dominant species. For each treatment, anoxic mud was spread evenly to a depth of 4 cm on top of each treatment plot. The mud was taken from areas adjacent to the plots and was obtained by scraping off the surface oxic layer and digging up the underlying mud from approximately 20 cm depth. Plots were subject to either low intensity treatments (burial every four weeks) or high (burial every two weeks). The experiment was carried out for 10 months at Creeksea and a year at Blackness. At

Creeksea numbers of *Tubificoides benedii* increased in both burial treatments until the third month (high burial) and sixth month (low burial). At Blackness increased numbers of *Tubificoides benedii* were found in both burial treatments after one month (Whomersley *et al*., 2010).

Laboratory experiments have shown that the snail *Hydrobia ulvae* can rapidly resurface through 5cm thick fine deposits, although this ability is reduced where deposited sediments contain little water (Chandrasekara & Frid, 1998). Field experiments where 10 cm of sediment were placed on intertidal sediments to investigate the effects of the beneficial use of dredged materials found that the abundance of *Hydrobia ulvae* had returned to ambient levels within 1 week (Bolam *et al*., 2004). The amphipod *Corophium volutator* may be sensitive to deposits at the pressure benchmark. Experimental fences placed on mudflats that caused sedimentation rates of 2-2.5 cm/month and reduced *Corophium volutator* densities from approximately 1700 m▯ to approximately 400 m▯. In areas without fences, *Corophium volutator* numbers increased from approximately 1700 per m▯ to 3500 per m▯ (Turk & Risk, 1981). In intertidal mudflats with similar characterizing species, experiments testing the effects of deposition of sediments typical of beach recharge, have found that recovery of biological assemblages is complete within two years (Bolam & Whomersley, 2003).

Sensitivity assessment. As the exposure to the pressure is for a single discrete event resistance is assessed as '**High**', resilience is also '**High**' and sensitivity is assessed as '**Not Sensitive**'. Confidence in this assessment is lower as the assessment is based on traits of the species characterizing the biotope and the relevant direct case studies present examples where impacts are not from single discrete events.

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

Studies have found that beach 'replenishment' or 'nourishment' that involves the addition of sediments on beaches can have several 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 from the natural habitat (Peterson *et al*., 2000).

Hediste diversicolor inhabits depositional environments. It is capable of burrowing to depths of up to 0.3 m and reworking sub-surface modifications of its burrow through fine clays and sand. Smith (1955) found no appreciable difference in the population of a *Hediste diversicolor* colony that had been covered by several inches of sand through which the worms tunnelled. Mobile polychaetes have been demonstrated to burrow through thick layers of deposits. Powilleit *et al*., (2009) studied the response of the polychaete *Nephtys hombergii* to smothering. This species successfully migrated to the surface of 32-41 cm deposited sediment layer of till or sand/till mixture and restored contact with the overlying water. While crawling upward to the new sediment surfaces burrowing velocities of up to 20 cm/day were recorded for *Nephtys hombergii*. Similarly, Bijkerk (1988, results cited from Essink 1999) indicated that the maximal overburden through which species could migrate was 60 cm through mud for *Nephtys* and 90 cm through sand. No further information was available on the rates of survivorship or the time taken to reach the surface.

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Sensitivity assessment. Deposition of up to 30 cm of fine material is likely to provide different impacts for the different species characterizing the biotope. Overall, although the characterizing species have some resistance to this to this pressure, populations are likely to be reduced. Resistance to initial smothering is '**Low'** Resilience is '**High'** 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

Litter, in the form of cigarette butts, has been shown to have an impact on ragworms. *Hediste diversicolor* showed increased burrowing times, 30% weight loss and a >2 fold increase in DNA damage when exposed to water with toxicants (present in cigarette butts) in quantities 60 fold lower than reported from urban run-off (Wright *et al.*, 2015). Studies are limited on impacts of litter on infauna and this UK study suggests health of infauna populations may be negatively impacted by this pressure.

Studies of sediment-dwelling, subsurface deposit feeding worms, a trait shared by species abundant in this biotope, showed negative impacts from ingestion of microplastics. For instance, *Arenicola marina* ingests micro-plastics that are present within the sediment it feeds within. Wright *et al.* (2013) carried out a lab study that displayed the presence of micro-plastics (5% UPVC) 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 the colonization of sediment by other species, reducing diversity in the biotopes the species occurs within. Wright *et al.* (2013) also present a case study based on their results, that in the intertidal regions of the Wadden Sea, where *Arenicola marina* is an important ecosystem engineer, *Arenicola marina* could ingest 33m of micro-plastics a year.

Sensitivity assessment. Impacts from the pressure 'litter' would depend upon the exact form of litter or man-made object introduced. Marine litter in the form of cigarette butts or microplastics may impact the health of populations of characterizing species. Significant impacts have been shown in laboratory studies but impacts at biotope scales are still unknown. Evidence and confidence in the assessment are limited and this pressure is **'Not assessed'**.

Electromagnetic changes	No evidence (NEv) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	No evidence (NEv) Q: NR A: NR C: NR

There is **no evidence** on effects of electric and magnetic fields on the characterizing species.

Electric and magnetic fields generated by sources such as marine renewable energy device/array cables may alter the behaviour of predators and affect infauna populations. Evidence is limited and occurs for electric and magnetic fields below the benchmark levels, confidence in the 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.* 2011). 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 species, which are predators of many polychaete species, including dab *Limanda limanda* and sole *Solea solea* 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.

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 and in response may retreat into the sediment for protection. 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

There is no direct evidence of the effects of changes in incident light on the characterizing species of this biotope. All characterizing species live in the sediment and do not rely on light levels directly to feed or find prey so that limited direct impact is expected. As this biotope is not characterized by the presence of primary producers it is not considered that shading would alter the character of the habitat directly.

More general changes to the productivity of the biotope may, however, occur. Beneath shading structures, there may be changes in microphytobenthos abundance. Littoral mud and sand support microphytobenthos on the sediment surface and within the sediment. The microphytobenthos consists of unicellular eukaryotic algae and cyanobacteria that grow within the upper several millimetres of illuminated sediments, typically appearing only as a subtle brownish or greenish shading. Mucilaginous secretions produced by these algae may stabilise fine substrata (Tait & Dipper, 1998). Shading will prevent photosynthesis leading to death or migration of sediment microalgae altering sediment cohesion and food supply to higher trophic levels. The impact of these indirect effects is difficult to quantify.

Sensitivity assessment. Based on the direct impact, biotope resistance is assessed as '**High**' and resilience is assessed as '**High**' (by default) and the biotope is considered to be '**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

Barriers that reduce the degree of tidal excursion may alter larval supply to suitable habitats from source populations. Barriers may also act as stepping stones for larval supply over greater distances (Adams *et al.*, 2014). Conversely, the presence of barriers at brackish waters may enhance local population supply by preventing the loss of larvae from enclosed habitats to environments, which are unfavourable, reducing settlement outside of the population. If a barrier (such as a tidal barrier) incorporated renewable energy devices such as tidal energy turbines, these devices may affect hydrodynamics and so migration pathways for larvae into and out of the biotope (Adams *et al*., 2014). However, evidence on this pressure is limited.

Sensitivity assessment. Resistance to this pressure is assessed as '**High**' and resilience as '**High**' by default. This biotope is, therefore, 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 interaction with bottom towed fishing gears and moorings are 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 and have limited, visual perception, this pressure is therefore 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

The 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

Intertidal mixed sediments may be colonized by a number of invasive non-indigenous species. Invasive species that alter the character of the biotope or that predate on characterizing species are most likely to result in significant impacts. Intertidal flats may be colonized by the invasive non-indigenous species *Crepidula fornicata* and the Pacific oyster *Magallana gigas*. 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 and food chain changes (OSPAR, 2009).

In the Wadden Sea, the Pacific oyster *Magallana gigas* has colonized intertidal flats (Smaal *et al*., 2005). This species consumes pelagic larvae reducing recruitment (Smaal *et al*., 2005). The most severe effects are likely to occur from impacts on sediment, where *Magallana gigas* create reefs on sedimentary flats that will prevent recruitment of juveniles and will restrict access of infauna to the sediment-water interface impacting respiration and feeding of bivalves such as *Limecola balthica* and *Scrobicularia plana* and polychaetes such as *Pygospio elegans* and disturbing the amphipod *Corophium volutator*. Burrowing infauna such as *Hediste diversicolor* and oligochaetes may persist within sediments but the overall character of the mixed sediment biotope would be altered.

The Manila clam (*Tapes philippinarium*), which was introduced to Poole Harbour for aquaculture in 1998, has become a naturalised population on the intertidal mudflats (occurring at densities of 60 clams/m^2 in some locations within the harbour (Jensen *et al*. 2007, cited in Caldow *et al*. 2007). Densities of *Cerastoderma edule* and *Abra tenuis* increased following the introduction of the Manila clam although the abundance of *Scrobicularia plana* and *Limecola balthica* declined (Caldow *et al*., 2005), although the decline of these species may have been caused by tri-butyl tin pollution (Langston *et al*., 2003) and may have facilitated the naturalization of the Manila clam.

The predatory veined whelk (*Rapana venosa*) and *Hemigrapsus takinei* are not established in the UK (although *Hemigrapsus takinei* has been recorded at two locations) could become significant predators of *Cerastoderma edule* and other species associated with the biotope in the future.

Sensitivity assessment. Intertidal muddy sands may be exposed to invasive species which can alter the character of the habitat (primarily *Crepidula fornicata* at the sublittoral fringe and *Magallana gigas*) leading to re-classification of this biotope, the biotope is considered to have **'Low'** resistance and **'Very low'** recovery (unless invasive species are removed). Biotope sensitivity is, therefore, assessed as **'High'.**

Introduction of microbial pathogens	High Q: Low A: NR C: NR	High Q: High A: High C: High	Not sensitive Q: Low A: Low C: Low

Limited evidence was returned by searches on the effect on *Hediste diversicolor* of the introduction of relevant microbial pathogens or metazoan disease vectors to an area where they are currently not present. Desrina *et al.* (2014) failed to induce infection of the 'White Spot Shrimp' virus in *Hediste diversicolor* by both feeding and immersion; a virus that can cause large scale mortality in shrimp. Marine oligochaetes host numerous protozoan parasites without apparent pathogenic effects even at high infestation levels (Giere & Pfannkuche, 1982).

Sensitivity assessment. Evidence and so confidence is limited, resistance is assessed as **'High'**, resilience is assessed as '**High**' and sensitivity is, therefore assessed as '**Not sensitive'**.

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

The sedimentary biotope and characterizing and associated species may be disturbed and damaged by static or mobile gears that are targeting other species. These direct, physical impacts are assessed through the abrasion and penetration of the seabed pressures. The sensitivity assessment for this pressure considers any biological/ecological effects resulting from the removal of target species on this biotope. Ragworms *Hediste diversicolor* are targeted by recreational and commercial bait diggers. The extent of the impact will depend on the fishing/removal method and spatial extent.

Populations of *Hediste diversicolor* are dominated by females; males may constitute up to 40% of the population but several reports suggest that the proportion of males is frequently lower (< 20%) (see Clay, 1967c). The sexes are externally indistinguishable except when approaching maturation and during spawning (see reproduction and adult general biology). Consequently, extraction e.g. by bait digging, of 50% of the specimens from within an area is likely to remove more females than males. A reduction in the female proportion of the population prior to spawning could reduce recruitment to the population. The mechanical action of the digging, even if the worms were not actually taken, may also cause some damage to the bodies. Recovery is dependent on the reproductive success and survival of the remaining population and colonization by adults from unaffected areas.

Sensitivity assessment. The key, characterizing species *Hediste diversicolor* may be targeted and their removal will alter the character of the biotope. Due to potential impacts on *Hediste diversicolor* populations, in particular females, biotope resistance is assessed as **'Low'**. Biotope resilience is assessed as **'High'** so that biotope sensitivity is assessed as **'Low'**.

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

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 these biotopes, 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). Loss of these species would alter the character of the biotope resulting in re-classification, and result in the loss of the ecosystem functions such as secondary production performed by these species.

Populations of oligochaetes provide food for macroinvertebrates fish and birds. For example, up to 67% of flounder and plaice stomachs examined from the Medway estuary (UK) (Van den Broek, 1978) contained the remains of *Tubificoides benedii (*studied as *Peloscolex benedeni)* and shrimps which in turn support higher trophic levels (predatory birds and fish). For some migratory birds, the characterizing species *Tubificoides benedii* can form an important part of the diet during winter (Bagheri & McLusky, 1984). Polychaetes and crustaceans are also predators of oligochaetes and may significantly reduce numbers (Giere & Pfannkuche, 1982 and references therein). The loss of the oligochaete population could, therefore, impact other trophic levels.

A dense field of *Corophium volutator* in the Wadden Sea disappeared completely during spring 1990 as a result of an epizootic by trematodes. The collapse of its population led to drastic changes in erosion patterns, sediment characteristics, and microtopography, as well as marked changes in the abundance of other macrofaunal species in the mudflat (Poulin & Mouritsen, 2006).

Sensitivity assessment. Removal of the characterizing species would result in the biotope being lost or reclassified. Thus, the biotope is considered to have a resistance of **'Low'** to this pressure and to have **'High'** resilience, so that sensitivity is assessed as **'Low'**.

Bibliography

Aberson, M., Bolam, S. & Hughes, R., 2016. The effect of sewage pollution on the feeding behaviour and diet of *Hediste* (*Nereis diversicolor* (OF Müller, 1776)) in three estuaries in south-east England, with implications for saltmarsh erosion. *Marine Pollution Bulletin*, **105** (1), 150-160.

Adams, T.P., Miller, R.G., Aleynik, D. & Burrows, M.T., 2014. Offshore marine renewable energy devices as stepping stones across biogeographical boundaries. *Journal of Applied Ecology*, **51** (2), 330-338.

Armonies, W., Herre, E. & Sturm, M., 2001. Effects of the severe winter 1995 / 1996 on the benthic macrofauna of the Wadden Sea and the coastal North Sea near the island of Sylt. *Helgoland Marine Research*, **55**, 170-175.

Attrill, M.J. ed., 1998. *A rehabilitated estuarine ecosystem: The environment and ecology of the Thames estuary*. Berlin: Springer Science & Business Media.

Bagheri, E. & McLusky, D., 1982. Population dynamics of oligochaetes and small polychaetes in the polluted forth estury ecosystem. *Netherlands Journal of Sea Research*, **16**, 55-66.

Bagheri, E.A. & McLusky, D.S., 1984. The oxygen consumption of *Tubificoides benedeni* (Udekem) in relation to temperature and its application to production biology. *Journal of Experimental Marine Biology and Ecology*, **78**, 187-197.

Bamber, R.N. & Spencer, J.F. 1984. The benthos of a coastal power station thermal discharge canal. *Journal of the Marine Biological Association of the United Kingdom*, **64**, 603-623.

Barnes, R.S.K., 1994. *The brackish-water fauna of northwestern Europe.* Cambridge: Cambridge University Press.

Bartels-Hardege, H.D. & Zeeck, E., 1990. Reproductive behaviour of *Nereis diversicolor* (Annelida: Polychaeta). *Marine Biology*, **106**, 409-412.

Bat, L., Raffaelli, D. & Marr, I.L., 1998. The accumulation of copper, zinc and cadmium by the amphipod *Corophium volutator* (Pallas). *Journal of Experimental Marine Biology and Ecology*, **223**, 167-184.

Beasley, T.M. & Fowler, S.W., 1976. Plutonium and Americium: uptake from contaminated sediments by the polychaete *Nereis diversicolor*. *Marine Biology*, **38**, 95-100.

Berthet, B., Mouneyrac, C., Amiard, C.J., Amiard-Triquet, C., Berthelot, Y., Le Hen, A., Mastain, O., Rainbow, S.P. & Smith, D.B., 2003. Accumulation and Soluble Binding of Cadmium, Copper, and Zinc in the Polychaete *Hediste diversicolor* from Coastal Sites with Different Trace Metal Bioavailabilities. *Archives of Environmental Contamination and Toxicology*, **45** (4), 468-478.

Beukema, J.J., 1989. Long term changes in macrozoobenthic abundance on the tidal flats of the western part of the Dutch Wadden Sea. *Helgolander Meeresuntersuchungen*, **43** (3-4), 405-415.

Beukema, J.J., 1990. Expected effects of changes in winter temperatures on benthic animals living in soft sediments in coastal North Sea areas. In *Expected effects of climatic change on marine coastal ecosystems* (ed. J.J. Beukema, W.J. Wolff & J.J.W.M. Brouns), pp. 83-92. Dordrecht: Kluwer Academic Publ.

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

Birtwell, I.K. & Arthur, D.R., 1980. The ecology of tubificids in the Thames Estuary with particular reference to *Tubifex costatus* (Claparède). In *Proceedings of the first international symposium on aquatic oligochaete biology, Sydney, British Colombia, Canada, May 1-4, 1979. Aquatic oligochaete biology* (ed. R.O. Brinkhurst & D.G. Cook), pp. 331-382. New York: Plenum Press

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

Bolam, S. & Whomersley, P., 2003. Invertebrate recolonization of fine-grained beneficial use schemes: An example from the southeast coast of England. *Journal of Coastal Conservation*, **9** (2), 159-169.

Bolam, S., Whomersley, P. & Schratzberger, M., 2004. Macrofaunal recolonization on intertidal mudflats: effect of sediment organic and sand content. *Journal of Experimental Marine Biology and Ecology*, **306** (2), 157-180.

Bolam, S.G., 2011. Burial survival of benthic macrofauna following deposition of simulated dredged material. *Environmental Monitoring and Assessment*, **181** (1-4), 13-27.

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.

Bouchet, V.M. & Sauriau, P.-G., 2008. Influence of oyster culture practices and environmental conditions on the ecological status of intertidal mudflats in the Pertuis Charentais (SW France): A multi-index approach. *Marine Pollution Bulletin*, **56** (11), 1898-1912.

Brils, J.M., Huwer, S.L., Kater, B.J., Schout, P.G., Harmsen, J., Delvigne, G.A.L. & Scholten, M.C.T., 2002. Oil effect in freshly spiked marine sediment on *Vibrio fischeri*, *Corophium volutator*, and *Echinocardium caudatum*. *Environmental Toxicology and Chemistry*, **21**, 2242-2251.

Brown, R.J., Conradi, M. & Depledge, M.H., 1999. Long-term exposure to 4-nonylphenol affects sexual differentiation and growth of the amphipod *Corophium volutator* (Pallas, 1766). *Science of the Total Environment*, **233**, 77-88.

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

Bryan G.W. & Hummerstone, L.G., 1971. Adaptation of the polychaete *Nereis diversicolor* to estuarine sediments containing high concentrations of heavy metals. I. General observations and adaption to copper. *Journal of the Marine Biological Association of the United Kingdom*, **51**, 845-863.

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

Bryant, V., McLusky, D.S., Roddie, K. & Newbery, D.M., 1984. Effect of temperature and salinity on the toxicity of chromium to three estuarine invertebrates (*Corophium volutator*, *Macoma balthica*, *Nereis diversicolor*). *Marine Ecology Progress Series*, **20**, 137-149.

Bryant, V., Newbery, D.M., McLusky, D.S. & Campbell, R., 1985. Effect of temperature and salinity on the toxicity of arsenic to three estuarine invertebrates (*Corophium volutator*, *Macoma balthica*, *Tubifex costatus*). *Marine Ecology Progress Series*, **24**, 129-137.

Bryant, V., Newbery, D.M., McLusky, D.S. & Campbell, R., 1985a. Effect of temperature and salinity on the toxicity of nickel and zinc to two estuarine invertebrates (*Corophium volutator*, *Macoma balthica*). *Marine Ecology Progress Series*, **24**, 139-153.

Buffet, P.-E., Poirier, L., Zalouk-Vergnoux, A., Lopes, C., Amiard, J.-C., Gaudin, P., Risso-de Faverney, C., Guibbolini, M., Gilliland, D., Perrein-Ettajani, H., Valsami-Jones, E. & Mouneyrac, C., 2014. Biochemical and behavioural responses of the marine polychaete Hediste diversicolor to cadmium sulfide quantum dots (CdS QDs): Waterborne and dietary exposure. *Chemosphere*, **100**, 63-70.

Caldow, R., McGrorty, S.,West, A., Durell, S. E. A. le V. dit, Stillman, R.,Anderson, S. 2005. Macro-invertebrate fauna in the intertidal mudflats. In *The ecology of Poole Harbour* (eds J. Humphreys & V. May), pp. 91–108. Amsterdam, The Netherlands: Elsevier B.V.

Caldow, R.W.G., Stillman, R.A., le V. dit Durell, S.E.A., West, A.D., McGrorty, S., Goss-Custard, J.D., Wood, P.J. & Humphreys, J., 2007. Benefits to shorebirds from invasion of a non-native shellfish. *Proceedings of the Royal Society, B*. **274**, 1449 – 1455.

Chandrasekara, W.U. & Frid, C.L.J., 1998. A laboratory assessment of the survival and vertical movement of two epibenthic gastropod species, *Hydrobia ulvae*, (Pennant) and *Littorina littorea* (Linnaeus), after burial in sediment. *Journal of Experimental Marine Biology and Ecology*, **221**, 191-207.

Chu, J.W. & Levin, L.A., 1989. Chu, J.W. & Levin, L.A., 1989. Photoperiod and temperature regulation of growth and reproduction in *Streblospio benedicti* (Polychaeta: Spionidae). *Invertebrate Reproduction & Development*, **15** (2), 131-142.

Clay, E., 1967c. *Literature survey of the common fauna of estuaries, 1.* Cirratulus cirratus*O.F. Müller. Imperial Chemical Industries Limited, Brixham Laboratory*, PVM45/A/374.

Coates, D.A., Deschutter, Y., Vincx, M. & Vanaverbeke, J., 2014. Enrichment and shifts in macrobenthic assemblages in an offshore wind farm area in the Belgian part of the North Sea. *Marine Environmental Research*, **95**, 1-12.

Collier, L.M. & Pinn, E.H., 1998. An assessment of the acute impact of the sea lice treatment Ivermectin on a benthic community. *Journal of Experimental Marine Biology and Ecology*, **230**, 131-147.

Conde, A., Novais, J.M. & Domínguez, J., 2013. Characterization of an estuarine environment by means of an index based on intertidal macrofauna. *Marine Pollution Bulletin*, **71** (1–2), 129-138.

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/

Conradi, M. & Depledge, M.H., 1999. Effects of zinc on the life-cycle, growth and reproduction of the marine amphipod *Corophium volutator*. *Marine Ecology Progress Series*, **176**, 131-138.

Craig, N.C.D. & Caunter, J.E., 1990. The effects of polydimethylsiloxane (PDMS) in sediment on the polychaete worm *Nereis diversicolor*. *Chemosphere*, **21**, 751-759.

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

Dales, R. P., 1950. The reproduction and larval development of *Nereis diversicolor* O. F. Müller. *Journal of the Marine Biological Association of the United Kingdom*, **29**, 321-360.

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

Davey, J.T. & George, C.L., 1986. Specific interactions in soft sediments: factors in the distribution of *Nereis (Hediste) diversicolor* in the Tamar Estuary. *Ophelia*, **26**, 151-164.

Davies, I.M, Gillibrand, P.A., McHenery, J.G. & Rae, G.H., 1998. Environmental risk of Ivermectin to sediment dwelling organisms. *Aquaculture*, **163**, 29-46.

Dedeh, A., Ciutat, A., Tran, D. & Bourdineaud, J.-P., 2014. DNA Alterations Triggered by Environmentally Relevant Polymetallic Concentrations in Marine Clams *Ruditapes philippinarum* and Polychaete Worms *Hediste diversicolor*. *Archives of Environmental Contamination and Toxicology*, **67**, (4), 651-658.

Desrina, Verreth, J.A.J., Vlak, J.M. & Verdegem, M.C.J., 2014. *Hediste diversicolor* (O.F. Mueller 1776) as a possible model to study White Spot Syndrome virus infection in polychaetes. *Asian Fisheries Science*, **27** (3), 199-211.

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.

Dolbeth, M., Cardoso, P.G., Grilo, T.F., Bordalo, M.D., Raffaelli, D. & Pardal, M.A., 2011. Long-term changes in the production by estuarine macrobenthos affected by multiple stressors. *Estuarine Coastal and Shelf Science*, **92** (1), 10-18.

Dries, R.R. & Theede, H., 1974. Sauerstoffmangelresistenz mariner Bodenvertebraten aus der West-lichen Ostsee. *Marine Biology*, **25**, 327-233.

Drolet, D., Kennedy, K. & Barbeau, M.A., 2013. Winter population dynamics and survival strategies of the intertidal mudflat amphipod *Corophium volutator* (Pallas). *Journal of Experimental Marine Biology and Ecology***, 441,** 126-137.

Esselink, P. & Zwarts, L., 1989. Seasonal trend in burrow depth and tidal variation in feeding activity of *Nereis diversicolor*. *Marine Ecology Progress Series*, **56**, 243-254.

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

Fernandez T.V. & Jones, N.V., 1990. The influence of salinity and temperature on the toxicity of zinc to *Nereis diversicolor*. *Tropical Ecology*, **31**, 40-46.

Fish, J.D. & Mills, A., 1979. The reproductive biology of *Corophium volutator* and *C. arenarium* (Crustacea: Amphipoda). *Journal of the Marine Biological Association of the United Kingdom*, **59**, 355-368.

Flach, E.C., 1993. The distribution of the amphipod *Corophium arenarium* in the Dutch Wadden Sea- relationships with sediment composition and the presence of cockles and lugworms. *Netherlands Journal of Sea Research*, **31** (3), 281-290.

Flach, E.C. & De Bruin, W., 1993. Effects of *Arenicola marina* and *Cerastoderma edule* on distribution, abundance and population structure of *Corophium volutator* in Gullmarsfjorden western Sweden. *Sarsia*, **78**, 105-118.

Flach, E.C. & De Bruin, W., 1994. Does the activity of cockles, *Cerastoderma edule* (L.) and lugworms, *Arenicola marina* (L.), make *Corophium volutator* Pallas more vulnerable to epibenthic predators: a case of interaction modification? *Journal of Experimental Marine Biology and Ecology*, **182**, 265-285.

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

Forbes, M.R., Boates, S.J., McNeil, N.L. & Brison, A.E., 1996. Mate searching by males of the intertidal amphipod *Corophium volutator* (Pallas). *Canadian Journal of Zoology*, **74**, 1479-1484.

Gamble, J., 1970. Anaerobic survival of the crustaceans *Corophium volutator*, *C. arenarium* and *Tanais chevreuxi*. *Journal of the Marine Biological Association of the United Kingdom*, **50** (03), 657-671.

Gamenick, I., Jahn, A., Vopel, K. & Giere, O., 1996. Hypoxia and sulphide as structuring factors in a macrozoobenthic community on the Baltic Sea shore: Colonization studies and tolerance experiments. *Marine Ecology Progress Series*, **144**, 73-85.

Gerdol, V. & Hughes, R.G., 1993. Effect of the amphipod *Corophium volutator* on the colonisation of mud by the halophyte *Salicornia europea*. *Marine Ecology Progress Series*, **97**, 61-69.

Germain, P., Miramand, P. & Masson, M., 1984. Experimental study of long-lived radionuclide transfers (americium, plutonium, technetium) between labelled sediments and annelidae (*Nereis diversicolor*, *Arenicola marina*). In *International symposium on the behaviour of long-lived radionuclides in the marine environment*, (ed. A.Cigna & C. Myttenaere), pp. 327-341. Luxembourg: Office for Official Publications of the European Communities.

Giere, O., 2006. Ecology and biology of marine oligochaeta–an inventory rather than another review. *Hydrobiologia*, **564** (1), 103-116.

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

Giere, O. & Rhode, B., 1987. Anatomy and ultrastructure of the marine oligochaete *Tubificoides benedii* (Tubificidae), with emphasis on its epidermis-cuticle-complex. In Brinkhurst, R.O. and Diaz, R.J. (eds.). *Aquatic Oligochaeta: Proceedings of the Third International Symposium on Aquatic Oligochaeta* held in Hamburg, Germany September 29-October 4, 1985, Dordrecht: Springer Netherlands, pp. 159-159.

Giere, O., Preusse, J. & Dubilier, N. 1999. *Tubificoides benedii* (Tubificidae, Oligochaeta) - a pioneer in hypoxic and sulfide environments. An overview of adaptive pathways. *Hydrobiologia*, **406**, 235-241.

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**.

Gillett, D.J., Holland, A.F. & Sanger, D.M., 2007. On the ecology of oligochaetes: monthly variation of community composition and environmental characteristics in two South Carolina tidal creeks. *Estuaries and Coasts*, **30** (2), 238-252.

Gogina, M., Glockzin, M. & Zettler, M.L., 2010a. Distribution of benthic macrofaunal communities in the western Baltic Sea with regard to near-bottom environmental parameters. 1. Causal analysis. *Journal of Marine Systems*, **79** (1), 112-123.

Goulletquer, P. & Heral, M., 1997. Marine molluscan production trends in France: from fisheries to aquaculture. *NOAA Tech. Rep. NMFS*, **129**.

Gray, J.S., 1971. The effects of pollution on sand meiofauna communities. *Thalassia Jugoslovica*, **7**, 76-86.

Gray, J.S., Clarke, K.R., Warwick, R.M. & Hobbs, G., 1990. Detection of initial effects of pollution on marine benthos - an example from the Ekofisk and Eldfisk oilfields, North Sea. *Marine Ecology Progress Series*, **66** (3), 285-299.

Hailey, N., 1995. *Likely impacts of oil and gas activities on the marine environment and integration of environmental considerations in* *licensing policy*. English Nature Research Report, no 145., Peterborough: English Nature.

Harris, G.J. & Morgan, E., 1984a. The effects of salinity changes on the endogenous circa-tidal rhythm of the amphipod *Corophium volutator* (Pallas). *Marine Behaviour and Physiology*, **10**, 199-217.

Harris, G.J. & Morgan, E., 1984b. The effects of ethanol, valinomycin and cycloheximide on the endogenous circa-tidal rhythm of the estuarine amphipod *Corophium volutator* (Pallas). *Marine Behaviour and Physiology*, **10**, 219-233.

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

Holmström, W.F. & Morgan, E., 1983b. The effects of low temperature pulses in rephasing the endogenous activity rhythm of *Corophium volutator* (Pallas). *Journal of the Marine Biological Association of the United Kingdom*, **63**, 851-860.

Hughes, R.G., 1988. Dispersal by benthic invertebrates: the *in situ* swimming behaviour of the amphipod *Corophium volutator*. *Journal of the Marine Biological Association of the United Kingdom*, **68**, 565-579.

Hunter, J., & Arthur, D.R., 1978. Some aspects of the ecology of *Peloscolex benedeni* Udekem (Oligochaeta: Tubificidae) in the Thames estuary. *Estuarine and Coastal Marine Science*, **6**, 197-208.

Hylleberg, J., 1975. The effect of salinity and temperature on egestion in mud snails (Gastropoda: Hydrobiidae). *Oecologia*, **21**, 279-289.

Jensen, K.T., 1992. Macrozoobenthos on an intertidal mudflat in the Danish Wadden Sea: Comparisons of surveys made in the 1930s, 1940s and 1980s. *Helgolander Meeresuntersuchungen*. Hamburg, **46** (4), 363-376.

Jensen, K.T. & Kristensen, L.D., 1990. A field experiment on competition between *Corophium volutator* (Pallas) and *Corophium arenarium* Crawford (Crustacea: Amphipoda): effects on survival, reproduction and recruitment. *Journal of Experimental Marine Biology and Ecology*, **137**, 1-24.

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/

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

Langston, W.J., Chesman, B.S., Burt, G.R., Hawkins, S.J., Readman, J. & Worsfold, P., 2003. Characterisation of European Marine Sites. Poole Harbour Special Protection Area. *Occasional Publication. Marine Biological Association of the United Kingdom*, **12**, 111.

Lardicci, C., Ceccherelli, G. & Rossi, F., 1997. *Streblospio shrubsolii* (Polychaeta : Spionidae): temporal fluctuations in size and reproductive activity. *Cahiers de Biologie Marine*, **38** (3), 207-214.

Levin, L. & Creed, E., 1986. Effect of temperature and food availability on reproductive responses of *Streblospio benedicti* (Polychaeta: Spionidae) with planktotrophic or lecithotrophic development. *Marine Biology*, **92** (1), 103-113.

Lewis, L., Davenport, J. & Kelly, T., 2002b. A study of the impact of a pipeline construction on estuarine benthic invertebrate communities. *Estuarine, Coastal and Shelf Science*, **55** (2), 213-221.

Lillicrap, A., Schaanning, M. & Macken, A., 2015. Assessment of the direct effects of biogenic and petrogenic activated carbon on benthic organisms. *Environmental Science & Technology*, **49** (6), 3705-3710.

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

Madsen, P.B. & Jensen, K., 1987. Population dynamics of *Macoma balthica* in the Danish Wadden Sea in an organically enriched area. *Ophelia*, **27**, 197-208.

McCabe, G.T. Jr., Hinton, S.A. & Emmett, R.L., 1998. Benthic invertebrates and sediment characteristics in a shallow navigation channel of the lower Columbia River. *Northwest Science*, **72**, 116-126.

McLusky, D., 1982. The impact of petrochemical effluent on the fauna of an intertidal estuarine mudflat. *Estuarine, Coastal and Shelf Science*, **14** (5), 489-499.

McLusky, D.S., 1968. Some effects of salinity on the distribution and abundance of *Corophium volutator* in the Ythan estuary. *Journal of the Marine Biological Association of the United Kingdom*, **48**, 443-454.

McLusky, D.S., Bryant, V. & Campbell, R., 1986. The effects of temperature and salinity on the toxicity of heavy metals to marine and estuarine invertebrates. *Oceanography and Marine Biology: an Annual Review*, **24**, 481-520.

McLusky, D.S., Teare, M. & Phizachlea, P., 1980. Effects of domestic and industrial pollution on distribution and abundance of aquatic oligochaetes in the Forth estuary. *Helgolander Wissenschaftliche Meeresuntersuchungen*, **33**, 384-392.

Meadows, P.S. & Ruagh, A.A., 1981. Temperature preferences and activity of *Corophium volutator* (Pallas) in a new choice apparatus. *Sarsia*, **66**, 67-72.

Mettam, C., Santhanam, V. & Havard, M.C.S., 1982. The oogenic cycle of *Nereis diversicolor* under natural conditions. *Journal of the Marine Biological Association of the United Kingdom*, **62**, 637-645.

Mills, A. & Fish, J., 1980. Effects of salinity and temperature on *Corophium volutator* and *C. arenarium* (Crustacea: Amphipoda), with particular reference to distribution. *Marine Biology*, **58** (2), 153-161.

Mouneyrac, C., Mastain, O., Amiard, J.C., Amiard-Triquet, C., Beaunier, P., Jeantet, A.-Y., Smith, B.D. & Rainbow, P.S., 2003. Trace-metal detoxification and tolerance of the estuarine worm Hediste diversicolor chronically exposed in their environment. *Marine*

Biology, **143** (4), 731-744.

Neal, K.J. & Avant, P. 2006. *Corophium volutator* A mud shrimp. In Tyler-Walters H. and Hiscock K. (eds) *Marine Life Information Network: Biology and Sensitivity Key Information Reviews*, [on-line]. Plymouth: Marine Biological Association of the United Kingdom. Available from: http://192.171.193.68/species/detail/1661

Normandeau, Exponent, T. Tricas, Gill, A., 2011. *Effects of EMFs from Undersea Power Cables on Elasmobranchs and Other Marine Species 2011*; U.S. Dept. of the Interior, Bureau of Ocean Energy Management, Regulation, and Enforcement, Pacific OCS Region, Camarillo, CA.OCS Study BOEMRE 2011-09.

Nugues, M., Kaiser, M., Spencer, B. & Edwards, D., 1996. Benthic community changes associated with intertidal oyster cultivation. *Aquaculture Research*, **27** (12), 913-924.

Olive, P.J.W. & Garwood, P.R., 1981. Gametogenic cycle and population structures of *Nereis (Hediste) diversicolor* and *Nereis (Nereis) pelagica* from North-East England. *Journal of the Marine Biological Association of the United Kingdom*, **61**, 193-213.

OSPAR Commission. 2009. Background document for *Modiolus modiolus* beds. *OSPAR Commission Biodiversity Series*. OSPAR Commission: London. Available from: http://www.ospar.org/documents?v=7193

Ozoh, P.T.E. & Jones, N.N., 1990. Capacity adaptation of *Hediste (Nereis) diversicolor* embryogenesis to salinity, temperature and copper. *Marine Environmental Research*, **29**, 227-243.

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.

Peterson, C.H., Bishop, M.J., Johnson, G.A., D'Anna, L.M. & Manning, L.M., 2006. Exploiting beach filling as an unaffordable experiment: benthic intertidal impacts propagating upwards to shorebirds. *Journal of Experimental Marine Biology and Ecology*, **338** (2), 205-221.

Peterson, C.H., Hickerson, D.H. & Johnson, G.G., 2000. Short-term consequences of nourishment and bulldozing on the dominant large invertebrates of a sandy beach. Journal of Coastal Research, 368-378.

Poulin, R. & Mouritsen, K.N., 2006. Climate change, parasitism and the structure of intertidal ecosystems. *Journal of Helminthology*, **80** (2), 183-192.

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

Raffaelli, D., 1999. Nutrient enrichment and trophic organisation in an estuarine food web. *Acta Oecologica*, **20** (4), 449-461.

Raffaelli, D., 2000. Trends in research on shallow water food webs. Journal of Experimental Marine Biology and *Ecology*, **250** (1-2), 223-232.

Raffaelli, D., Limia, J., Hull, S. & Pont, S., 1991. Interactions between the amphipod *Corophium volutator* and macroalgal mats on estuarine mudflats. *Journal of the Marine Biological Association of the United Kingdom*, **71**, 899-908.

Reise, K., Herre, E., & Sturm, M. 1989. Historical changes in the benthos of the Wadden Sea around the island of Sylt in the North Sea. *Helgoländer Meeresuntersuchungen*, **43**, 417-433.

Roddie, B., Kedwards, T., Ashby-Crane, R. & Crane, M., 1994. The toxicity to *Corophium volutator* (Pallas) of beach sand contaminated by a spillage of crude oil. *Chemosphere*, **29** (4), 719-727.

Rodrigues, A.M., Quintino, V., Sampaio, L., Freitas, R. & Neves, R., 2011. Benthic biodiversity patterns in Ria de Aveiro, Western Portugal: Environmental-biological relationships. *Estuarine, Coastal and Shelf Science*, **95** (2–3), 338-348.

Scaps, P., 2002. A review of the biology, ecology and potential use of the common ragworm *Hediste diversicolor* (O.F. Müller) (Annelida: Polychaeta). *Hydrobiologia*, **470**, 203-218.

Schueckel, U. & Kroencke, I., 2013. Temporal changes in intertidal macrofauna communities over eight decades: A result of eutrophication and climate change. *Estuarine Coastal and Shelf Science*, **117**, 210-218.

Sheehan, E.V., 2007. *Ecological impact of the Carcinus maenas (L.) fishery'crab-tiling'on estuarine fauna*. Ph.D. thesis, University of Plymouth.

Shepherd, P.C.F. & Boates, S.J., 1999. Effects of commercial baitworm harvest on semipalmated sandpipers and their prey in the Bay of Fundy hemispheric shorebird reserve. *Conservation Biology*, **13**, 347-356.

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

Smaal, A., van Stralen, M. & Craeymeersch, J., 2005. Does the introduction of the Pacific oyster *Crassostrea gigas* lead to species shifts in the Wadden Sea? The comparative roles of suspension-feeders in ecosystems: *Springer*, **47,** 277-289.

Smith, J.E., 1955. Salinity variation in interstitial water of sand at Kames Bay, Millport, with reference to the distribution of *Nereis diversicolor Journal of the Marine Biological Association of the United Kingdom*, **34**, 33-46.

Smith, J.E., 1964. On the early development of *Nereis diversicolor* in different salinities. *Journal of Morphology*, **114**, 437-464.

Smith, R.I., 1977. Physiological and reproductive adaptations of *Nereis diversicolor* to life in the Baltic Sea and adjacent waters. In *Essays on polychaetous annelids* (ed. D.J. Reish & R. Fauchald), pp. 373-390. Los Angeles: University of Southern California.

Stekoll, M.S., Clement, L.E. & Shaw, D.G., 1980. Sublethal effects of chronic oil exposure on the intertidal clam *Macoma balthica*. *Marine Biology*, **57**, 51-60.

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.

Theede, H., 1973. Comparative studies on the influence of oxygen deficiency and hydrogen sulphide on marine bottom invertebrates. *Netherlands Journal of Sea Research*, **7**, 244-252.

Theede, H., Schaudinn, J. & Saffè, F., 1973. Ecophysiological studies on four *Nereis* species in the Kiel Bay. *Oikos Supplementum*, **15**, 246-252,

Thiel, M., Stearns, L. & Watling, L., 1998. Effects of green algal mats on bivalves in a New England mud flat. *Helgoländer Meeresuntersuchungen*, **52** (1), 15-28.

Tillin, H.M., Hull, S.C. & Tyler-Walters, H., 2010. Development of a sensitivity matrix (pressures-MCZ/MPA features). *Report to the Department of the Environment, Food and Rural Affairs from ABPmer, Southampton and the Marine Life Information Network (MarLIN) Plymouth: Marine Biological Association of the UK.*, Defra Contract no. MB0102 Task 3A, Report no. 22., London, 145 pp.

Turk, T.R. & Risk, M.J., 1981. Invertebrate populations of Cobequid Bay, Bay of Fundy. *Canadian Journal of Fisheries and Aquatic Sciences*, **38**, 642-648.

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

Van den Broek, W., 1978. Dietary habits of fish populations in the Lower Medway Estuary. *Journal of Fish Biology*, **13** (5), 645-654.

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.

Verdonschot, P., Smies, M. & Sepers, A., 1982. The distribution of aquatic oligochaetes in brackish inland waters in the SW Netherlands. *Hydrobiologia*, **89** (1), 29-38.

Vismann, B., 1990. Sulphide detoxification and tolerance in *Nereis (Hediste) diversicolor* and *Nereis (Neanthes) virens* (Annelida: Polychaeta). *Marine Ecology Progress Series*, **59**, 229-238.

Volkenborn, N. & Reise, K., 2006. Lugworm exclusion experiment: Responses by deposit feeding worms to biogenic habitat transformations. *Journal of Experimental Marine Biology and Ecology*, **330** (1), 169-179.

Whomersley, P., Huxham, M., Bolam, S., Schratzberger, M., Augley, J. & Ridland, D., 2010. Response of intertidal macrofauna to multiple disturbance types and intensities – an experimental approach. *Marine Environmental Research*, **69** (5), 297-308.

Wilson, W.H. & Parker, K., 1996. The life history of the amphipod, *Corophium volutator*: the effects of temperature and shorebird predation. *Journal of Experimental Marine Biology and Ecology*, **196**, 239-250.

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., Reid, M.J., Thomas, K.V. & Galloway, T.S., 2015. Bioaccumulation and biological effects of cigarette litter in marine worms. *Scientific reports*, **5**, 14119.

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.

Zuhlke, R. & Reise, K., 1994. Response of macrofauna to drifting tidal sediments. *Helgolander Meeresuntersuchungen*, **48** (2-3), 277-289.