# *Hediste diversicolor* and *Limecola balthica* in littoral sandy mud

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

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2016-08-31

This review can be cited as:
Tillin, H.M. & Rayment, W., 2016. [Hediste diversicolor] and [Limecola balthica] in littoral sandy mud. 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.
DOI https://dx.doi.org/10.17031/marlinhab.209.1

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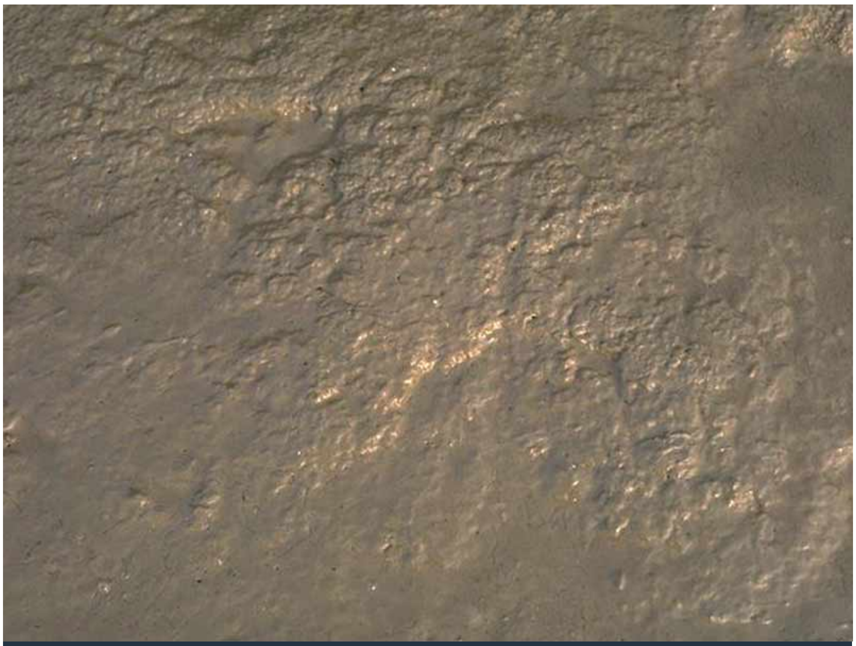

**Upper shore mud with *Hediste* holes and tracks.**
**Photographer:** Roger Covey
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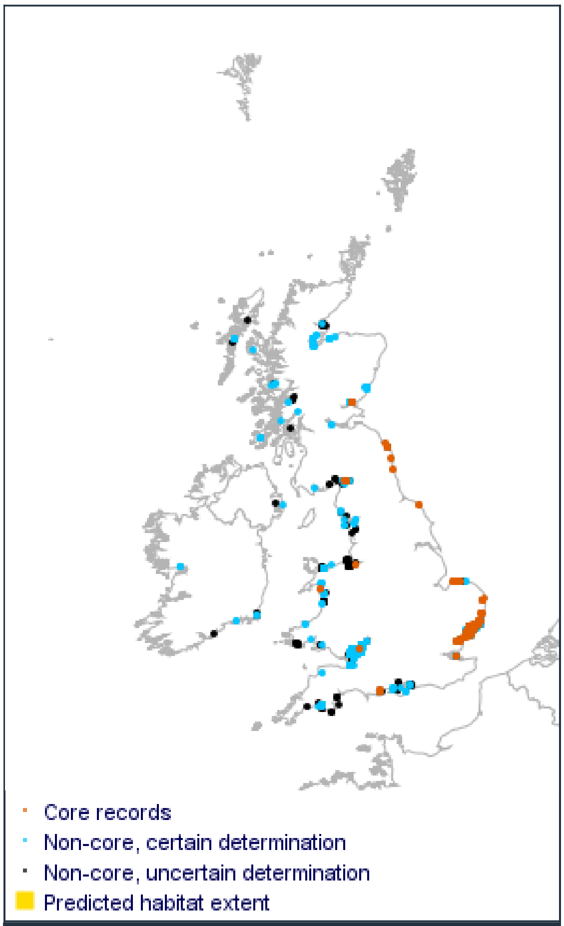

17-09-2018
Biotope distribution data provided by EMODnet Seabed Habitats (www.emodnet-seabedhabitats.eu)

**Researched by** Dr Heidi Tillin & Will Rayment **Refereed by** This information is not refereed.

# Summary

## UK and Ireland classification

| | | |
|---|---|---|
| **EUNIS 2008** | A2.312 | *Hediste diversicolor* and *Macoma balthica* in littoral sandy mud |
| **JNCC 2015** | LS.LMu.MEst.HedLim | *Hediste diversicolor* and *Limecola balthica* in littoral sandy mud |
| **JNCC 2004** | LS.LMu.MEst.HedMac | *Hediste diversicolor* and *Macoma balthica* in littoral sandy mud |
| **1997 Biotope** | LS.LMU.SMu.HedMac | *Hediste diversicolor* and *Macoma balthica* in sandy mud shores |

## Description

Mainly mid and lower shore sandy mud or mud in lower estuaries, sheltered bays and marine inlets, often subject to variable salinity. The main characterizing species are the ragworm *Hediste* *diversicolor*, the baltic tellin *Limecola balthica*, and the oligochaetes *Tubificoides benedii* and *Tubificoides pseudogaster*. Further polychaetes that are often common or abundant include *Pygospio elegans*, *Streblospio shrubsolii*, *Tharyx killariensis*, *Aphelochaeta marioni*, *Capitella capitata* and *Manayunkia aestuarina*. The oligochaete *Heterochaeta costata* and the mud shrimp *Corophium volutator* may be abundant. The spire shell *Hydrobia ulvae* is often common. Other species which occur in a significant proportion of samples include the polychaetes *Eteone longa* and *Nephtys hombergii*, and bivalves such as the cockle *Cerastoderma edule* and *Abra tenuis*. The sand gaper *Mya arenaria* is superabundant in about a quarter of the samples for this biotope. *Mya arenaria* is probably present in a higher proportion of areas of this biotope but may be missed in core samples due to its size. HedLim may occur on the mid/lower shore of lower estuarine shores, with HedLimEte or LimAre on the upper shore. HedLimScr, Hed.Str, NhomAph, and Hed.Cvol may be present on the same shore. *Ulva* spp. may form mats on the surface of the mud during the summer months, particularly in areas of nutrient enrichmnent. (Infromation from Connor *et al*., 2004; JNCC 2015).

## Depth range

Mid shore, Lower shore

## Additional information

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## Listed By

- none -

## Further information sources

Search on:

G G JNCC

# Habitat review

## Ecology

### Ecological and functional relationships

- The dominant and characterizing species in the biotope (*Limecola balthica* and *Hediste diversicolor*) are infaunal and display plasticity in their feeding methods (McLusky & Elliott, 1981; Nielsen *et al.*, 1995). They are primarily deposit feeders but are able to switch to suspension feeding when conditions allow. For example, in the case of *Hediste diversicolor*, if phytoplankton concentrations are sufficiently high the species switches from predatory and surface deposit-feeding to suspension feeding. This behaviour is maintained as long as the phytoplankton concentration remains above a 'trigger level' of 1-3 µg chlorophyll a per litre (reviewed by Riisgard, 1994).
- Obligate deposit feeders are also numerous in the biotope, e.g. *Pygospio elegans*, *Aphelochaeta marioni*, *Arenicola marina* and *Hydrobia ulvae*. Competition for resources is likely to occur between the deposit feeders. For example, densities of the amphipod, *Corophium volutator*, and the gastropod, *Hydrobia ulvae*, are strongly negatively correlated although the mechanism of the interaction is not well understood (Barnes & Hughes, 1992).
- Furthermore, *Corophium volutator* and the infaunal annelid species in the biotope probably interfere strongly with each other. Adult worms probably reduce amphipod numbers by disturbing their burrows, while high densities of amphipods can prevent establishment of worms by consuming larvae and juveniles (Olafsson & Persson, 1986). *Arenicola marina* has been shown to have a strong negative effect on *Corophium volutator* due to reworking of sediment causing the amphipod to emigrate (Flach, 1992), and also on *Pygospio elegans* (Reise, 1985).
- Suspension feeding bivalves , e.g. *Cerastoderma edule*, *Mya arenaria* and *Abra tenuis*, may occur where hydrodynamic conditions allow, i.e. in areas of stronger flow with coarser sediments.
- Important epibenthic predators in the biotope include the shore crab, *Carcinus maenas*, and shrimps, e.g. *Crangon crangon*, which take infaunal populations of small bivalves, polychaetes and crustaceans (Elliot *et al.*, 1998). *Carcinus maenas* has been shown to significantly reduce the numbers of *Manayunkia aestuarina* on mudflats (McClusky, 1989) and along with *Crangon crangon* may reduce the population of *Corophium volutator* in estuaries by more than 50% (Pihl, 1985).
- Intertidal mudflats are important feeding areas for many fish species. The most significant predators are probably the flatfish, including sole, *Solea solea*, dab, *Limanda limanda*, flounder, *Platichthys flesus*, and plaice, *Pleuronectes platessa*, which feed on polychaetes and their tails (e.g. of *Hediste diversicolor* and *Arenicola marina*), bivalve young and siphons (e.g. of *Limecola balthica*) and crustaceans (see review by Elliot *et al.*, 1998).
- Mobile epibenthic predators are probably responsible for preventing the infauna reaching carrying capacity. For example, exclusion experiments by Reise (1985) on intertidal mudflats showed small fish and decapod crustaceans can significantly reduce numbers of *Cerastoderma edule*, *Tubificoides benedii*, *Pygospio elegans*, *Aphelochaeta marioni*, *Eteone longa* and *Corophium volutator*.
- Carnivorous annelids, e.g. *Eteone longa* and *Nephtys hombergii*, operate at the trophic level below *Carcinus maenas* (Reise, 1985). They predate the smaller annelids and crustaceans in the biotope.

- Macroalgae, such as *Ulva lactuca* and *Ulva* sp., probably only occur in the biotope where suitable hard substrata exist for attachment. However, *Ulva lactuca* may grow free floating in very sheltered conditions.

### Seasonal and longer term change

Seasonal changes are likely to occur in the abundance of fauna in the biotope due to seasonal recruitment processes and variations in recruitment success. For example, in the case of *Limecola balthica*, Bonsdorff *et al.* (1995) reported juvenile density in the Baltic Sea following settlement in late summer to be 300,000/m$^2$ decreasing to a stable adult density of 1,000/m$^2$, and Ratcliffe *et al.* (1981) reported adult densities in the Humber Estuary, UK, between 5,000/m$^2$ and 40,000/m$^2$ depending on time since a successful spat fall. Variation in abundance is also very pronounced in the polychaete *Aphelochaeta marioni*. For example, in the Wadden Sea, peak abundance occurred in January (71,200 individuals per m$^2$) and minimum abundance occurred in July (22,500 individuals per m$^2$) following maximum spawning activity between May and July (Farke, 1979). However, the spawning period varies according to environmental conditions and so peak abundances will not necessarily occur at the same time each year. For example, Gibbs (1971) reported *Aphelochaeta marioni* spawning in late autumn in Stonehouse Pool, Plymouth Sound.

Some species make seasonal migrations in response to environmental conditions. For example, Beukema & De Vlas (1979) reported that 30% of the *Limecola balthica* population migrated into the subtidal during winter apparently in response to low temperatures. Migration is achieved by burrowing (Bonsdorff, 1984; Guenther, 1991) and/or floating (Sörlin, 1988) .

Macroalgal cover typically varies through the year due to changes in temperature and light availability. The green macroalgae in the biotope are likely to proliferate in the spring and summer and die back in the autumn and winter in conjunction with decreasing light levels and temperature and increased disturbance by storm events. Production by microphytobenthos and microalgae is also likely to be higher in spring and summer, increasing food availability for deposit feeders and suspension feeders in the biotope.

The biotope is likely to be susceptible to increased wave action during storms, particularly in winter. Storms may result in changes in sediment composition and washing out of infauna, leaving the biotope available for recolonization (see review by Hall, 1994). The infauna may be damaged by wave action, displaced from their preferred habitat and/or cast ashore, resulting in mortality. For example, Tamaki (1987) studied the passive transport by waves and tidal currents of the adults of 5 polychaete species. One species exhibited a landward shift in its centre of population during winter when the wave effects were most profound, and reoccupied its summer position by active migration of adults.

### Habitat structure and complexity

- The substratum in the biotope is uniform sediment with little structural diversity provided by either physiographic features or the biota. Some 3-dimensional structure is provided by the burrows of infauna such as *Arenicola marina*. Most species living within the sediment are limited to the area above the anoxic layer, the depth of which will vary depending on sediment particle size and organic content. However, the presence of burrows allows a larger surface area of sediment to become oxygenated, and thus enhances the survival of a considerable variety of small species (Pearson & Rosenberg, 1978).
- Reworking of sediments by deposit feeders increases bioturbation and potentially causes a change in the substratum characteristics and the associated community (e.g. Rhoads & Young, 1970). For example, Widdows *et al.* (1998) reported that typical abundances of

*Limecola balthica* increased sediment resuspension and/or erodability four fold and that there was a significant positive correlation between density of the species and sediment resuspension.

- Where present, the macroalgae provide some structural complexity in the habitat, providing cover and sites for attachment of epifauna and epiphytes.

### Productivity

- Primary production in the biotope comes from benthic microalgae and water column phytoplankton (Elliott *et al.*, 1998).
- Production by benthic unicellular and filamentous algae in the littoral zone accounts for 0.2-1.3 g C/m▯ daily, depending on water clarity (Barnes & Hughes, 1992). Benthic microalgae are able to photosynthesize over a much wider range of light intensity than the planktonic species. There is little or no photoinhibition, adapting littoral species to the full sunlight experienced at low tide while still utilizing the very low light intensities at high tide (Barnes & Hughes, 1992).
- Where present, macroalgae also contribute to primary production in the biotope. They exude considerable amounts of dissolved organic carbon which are taken up readily by bacteria and possibly by some larger invertebrates.
- The majority of nutrients in the biotope are derived from allochthonous sources. Mudflats receive large inputs of nutrients, sediment and organic matter from the sea and land discharges of river water and sewage, resulting in high productivity despite low levels of primary production (Elliott *et al.*, 1998).

### Recruitment processes

The bivalves that characterize the biotope are capable of high recruitment and rapid recovery. For example, adult *Limecola balthica* spawn at least once a year and are highly fecund (Caddy, 1967). There is a planktotrophic larval phase which lasts up to 2 months (Fish & Fish, 1996) and so dispersal over long distances is potentially possible given a suitable hydrographic regime. Following settlement, development is rapid and sexual maturity is attained within 2 years (Gilbert, 1978; Harvey & Vincent, 1989). In addition to larval dispersal, dispersal of juveniles and adults occurs via burrowing (Bonsdorff, 1984; Guenther, 1991), floating (Sörlin, 1988) and probably via bedload transport (Emerson & Grant, 1991). It is expected therefore that recruitment can occur from both local and distant populations.

The infaunal polychaetes *Hediste diversicolor*, Arenicola marina and *Aphelochaeta marioni* have high fecundity and the eggs develop lecithotrophically within the sediment or at the sediment surface (Farke, 1979; Beukema & de Vlas, 1979). There is no pelagic larval phase and the juveniles disperse by burrowing. Recruitment must occur from local populations or by longer distance dispersal of postlarvae in water currents or during periods of bedload transport. For example, Davey & George (1986), found evidence that larvae of *Hediste diversicolor* were tidally dispersed within the Tamar Estuary over a distance of 3 km, as larvae were found on an intertidal mudflat which previously lacked a resident population of adults. Recruitment is therefore likely to be predictable if local populations exist but patchy and sporadic otherwise.

The deposit feeding gastropod *Hydrobia ulvae* appears to display plasticity in its developmental mechanism. Fish & Fish (1996) report planktotrophic development with a free-swimming larval phase lasting 3 weeks, while Pilkington (1971) stated that development occurred via a non-feeding benthic larvae which metamorphosed in just 3 days. It is possible that *Hydrobia ulvae* is able to change its developmental mechanism according to environmental conditions. If conditions are favourable, the eggs may hatch, develop directly and recruit locally. In more stressful conditions, it may benefit the individual to disperse its offspring more widely via a planktotrophic larva. Recruitment of shallow burrowing infaunal species can depend on adult movement by bedload sediment transport and not just spat settlement and juvenile dispersal. Emerson & Grant (1991) investigated recruitment in *Mya arenaria* and found that bedload transport was positively correlated with clam transport. They concluded that clam transport at a high energy site accounted for large changes in clam density. Furthermore, clam transport was not restricted to storm events and the significance is not restricted to *Mya arenaria* recruitment. Many infauna, e.g. polychaetes, gastropods, nematodes and other bivalves, will be susceptible to movement of their substratum.

### Time for community to reach maturity

No information was found concerning time taken for the community to reach maturity. However, the characterizing species are highly fecund and quick to grow and mature and so the community would be expected to reach maturity within 5 years.

### Additional information

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## Preferences & Distribution

### Habitat preferences

| | |
|---|---|
| **Depth Range** | Mid shore, Lower shore |
| Water clarity preferences | |
| **Limiting Nutrients** | No information found |
| **Salinity preferences** | Full (30-40 psu), Variable (18-40 psu) |
| **Physiographic preferences** | Enclosed coast / Embayment |
| **Biological zone preferences** | Eulittoral |
| **Substratum/habitat preferences** | Mud, Sandy mud |
| **Tidal strength preferences** | |
| **Wave exposure preferences** | Extremely sheltered, Sheltered, Very sheltered |
| **Other preferences** | |

### Additional Information

## Species composition

### Species found especially in this biotope

### Rare or scarce species associated with this biotope

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## Additional information

# Sensitivity review

## Sensitivity characteristics of the habitat and relevant characteristic species

Connor *et al.* (2004) summarise the biotope as being located mainly in mid and lower shore sandy mud or mud in lower estuaries, sheltered bays and marine inlets, often subject to variable salinity. The main characterizing species are the ragworm *Hediste diversicolor*, the baltic tellin *Macoma balthica*, and the oligochaetes *Tubificoides benedii* and *Tubificoides pseudogaster*. Further polychaetes that are often common or abundant include *Pygospio elegans*, *Streblospio shrubsolii*, *Tharyx killariensis*, *Aphelochaeta marioni*, *Capitella capitata* and *Manayunkia aestuarina*. The oligochaete *Heterochaeta costata* and the mud shrimp *Corophium volutator* may be abundant. The spire shell *Hydrobia ulvae* is often common. Other species which occur in a significant proportion of samples include the polychaetes *Eteone longa* and *Nephtys hombergii*, and bivalves such as the cockle *Cerastoderma edule* and *Abra tenuis*. The sand gaper *Mya arenaria* is superabundant in about a quarter of the samples for this biotope. *Mya arenaria* is probably present in a higher proportion of areas of this biotope, but may be missed in core samples due to its size.

This review focuses on the ragworm *Hediste diversicolor* and the baltic tellin *Macoma balthica* as loss of these species would lead to loss of the biotope. Sandy mud or mud is also required to support these species and loss of these sediment types would also lead to loss of the biotope. Other species that are common or abundant are considered where they are likely to be affected by a pressure.

## Resilience and recovery rates of habitat

When impacted this biotope may recover through 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.

Generally *Hediste diversicolor* is reported to reach maturity between one and three years of age, like other Nereidae, *Hediste diversicolor* are monotelic, that is, they reproduce only once in their lifetime and then die (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 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 a 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.

The life history characteristics of □*Limecola balthica* give the species strong powers of recoverability. Adults spawn at least once a year and are highly fecund (Caddy, 1967). Females are capable of producing 10,000-30,000 eggs (MES, 2010). There is a planktotrophic larval phase which lasts up to 2 months (Fish & Fish, 1996) and so dispersal over long distances is potentially possible given a suitable hydrographic regime. Following settlement, development is rapid and sexual maturity is attained within 2 years (Gilbert, 1978; Harvey & Vincent, 1989). In addition to larval dispersal, dispersal of juveniles and adults occurs via burrowing (Bonsdorff, 1984; Guenther, 1991), floating (Sörlin, 1988) and probably via bedload transport (Emerson & Grant, 1991). It is expected therefore that recruitment can occur from both local and distant populations. Bonsdorff (1984) studied the recovery of a *Limecola balthica* (as *Macoma balthica*) population in a shallow, brackish bay in SW Finland following the removal of the substratum by dredging in the summer of 1976. Recolonization of the dredged area by *Limecola balthica* began immediately after the disturbance to the sediment and by November 1976, the *Limecola balthica* population had recovered to 51 individuals/m□. One year later there was no detectable difference in the *Limecola balthica* population between the recently dredged area and a reference area elsewhere in the bay. In 1976, two generations could be detected in the newly established population indicating that active immigration of adults was occurring in parallel to larval settlement. In 1977, up to six generations were identified, giving further evidence of active immigration to the dredged area.

A study by Verdelhos *et al*. (2011) suggested the existence of different life strategies within populations of *Scrobicularia plana* depending on temperature, latitudinal gradient and local habitat conditions. Higher-latitude populations usually exhibited low abundance values, shorter reproduction periods and lower growth rates. Conde *et al*. (2011) compared recruitment of the bivalve *Scrobicularia plana* to excavated and un-excavated control plots (expected to enhance the deposition of bivalve spat if the settlement of bivalves was the result of a passive process) at different shore levels in Portugal. Juveniles of both bivalve species were found to avoid excavated plots, showing significantly higher abundance in control plots. The data strongly suggested that recruited bivalves actively avoid unsuitable substrata.

The polychaete *Pygospio elegans* have life history strategies that allow rapid colonization and population increase in disturbed and defaunated patches where there is little competition from other species. *Pygospio elegans* exhibit a number of reproductive strategies (a trait known as poecilogony). Larvae may develop directly allowing rapid population increase in suitable patches or they may have a planktonic stage (allowing colonization of new habitats). Experimental defaunation studies have shown an increase in *Pygospio elegans*, higher than background abundances within 2 months, reaching maximum abundance within 100 days (Van Colen *et al*. 2008). Following a period of anoxia in the Bay of Somme (north France) that removed cockles, *Pygospio elegans* increased rapidly but then decreased as cockle abundance recovered and sediments were disturbed by cockle movement (Desprez *et al*., 1992). Re-colonization of *Pygospio elegans*, was observed in 2 weeks by Dittmann *et al*. (1999) following a 1 month long defaunation of the sediment. However, McLusky *et al.* (1983) found that *Pygospio elegans* were significantly depleted for >100 days after harvesting (surpassing the study monitoring timeline). Ferns *et al*. (2000) found that tractor-towed cockle harvesting removed 83% of *Pygospio elegans* (initial density 1850 per $m^2$). In muddy sand habitats, *Pygospio elegans* had not recovered their original abundance after 174 days (Ferns *et al*., 2000). These results are supported by work by Moore (1991) who also found that cockle dredging can result in reduced densities of some polychaete species, including *Pygospio elegans*. Rostron (1995) undertook experimental dredging of sandflats with a mechanical cockle dredger, including a site comprised of stable, poorly sorted fine sands with small pools and *Arenicola marina* casts with some algal growths. At this site, post-dredging, there was a decreased number of *Pygospio elegans* with no recovery to pre-dredging numbers after six months.

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 Fourth 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 and 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 southeast 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 prior to 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)* appeared in upper sediment layers in experimentally defaunated patches (4$m^2$) 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), Bolam & Whomersley, (2003), Bonsdorff (1984), and Gamenick *et al*. (1996). Resilience is assessed as 'Medium' (2-10 years) where, resistance is 'None' and habitat recovery may also be required.

**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<br>Q: High A: High C: High | High<br>Q: High A: High C: High | Not sensitive<br>Q: High A: High C: High |

*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 have an adverse 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* were not strongly affected by heat waves in an estuary in north western Portugal, where temperatures reached 40°C in intertidal pools (higher temperatures than experienced around UK and Irish coasts) (Dolbeth *et al.*, 2011). Grilo *et al.*, 2011) found that, at a Portuguese site, surface deposit feeders gradually decreased in periods of higher temperatures. However, sub-surface deposit feeders became dominant for up to 3 years after heat wave conditions had passed.

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.

In Europe, *Limecola balthica* occurs as far south as the Iberian Peninsula and hence would be expected to tolerate higher temperatures than experienced in Britain and Ireland. Oertzen (1969) recorded that *Limecola balthica* (as *Macoma balthica*) could tolerate temperatures up to 49°C before thermal numbing of gill cilia occurred, presumably resulting in death. Ratcliffe *et al.*, 1981) reported that *Limecola balthica* (as *Macoma balthica*) from the Humber Estuary, UK, tolerated 6 hours of exposure to temperatures up to 37.5°C with no mortality. Wilson (1981) show that the lethal temperatures for *Limecola balthica* change between seasons. Critical temperatures were studied for a *Limecola balthica* population in Dublin Bay, and a summer maximum of 37.5 and winter maximum of 27.5 were reported (Wilson, 1981). Tolerances were also reported to change with height up the shore, which suggested tolerances adapt to variable conditions.

Field observations and laboratory experiments showed that *Limecola balthica* spawns (criterion: 50% spent) in spring when the gradual increase of the mean (monthly averaged) water temperature surpasses 8.3°C. The success of spawning and recruitment is affected by the timing of the spring phytoplankton bloom and avoidance of the main settlement of the predator *Crangon crangon* on intertidal shores (Philippart *et al*., 2003). A mismatch in spawning cues due to an acute increase in temperature could result in low recruitment or recruitment failures

Despite apparent adaptation to regional temperature ranges, Barda *et al.* (2014) studied populations of *Limecola balthica* (as *Macoma balthica*) in the Baltic Sea and found that increased temperature reduced growth rates. Beukema *et al.* (2014) also warn that increasing water temperatures as a result of global warming are likely to shorten the growing season (typically late winter to early spring) if warmer spring and summer water temperatures are experienced. Jansen *et al.* (2007) suggest that temperature increases in the Spanish coast along the Bay of Biscay over the past 40 years caused loss of *Limecola balthica*populations, due to short-term but frequent exposure to >30° C in the Spanish estuaries, which induced elevated maintenance rates in *Limecola balthica*, and ultimately starvation. Repeated recruitment failure also occurred after mild winters in a comparable North Sea location, probably due to enhanced survival of predators(Beukema, 1992, Schueckel & Kroencke, 2013; Beukema *et al*., 2001). As a result, Jansen *et al.* (2007) predict the southern limit of the species will progressively shift north if temperatures continue to rise.

*Pygospio elegans* also shows a relationship between timing of reproduction and temperature. Gibson and Harvey (2000) in a study on asexual reproduction of *Pygospio elegans* in Nova Scotia, Canada found that temperature did not influence reproduction strategy (planktotrophy, lecithotrophy or asexual reproduction) but cite Anger (1984) that environmental conditions, including temperature influence timing of reproduction.

*Tubificoides benedii* increased in abundance in mudflat habitats in Jade Bay, 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.*Eteone longa* and *Pygospio elegans* were summer visitors at the same sites (Bamber & Spencer, 1984) and are considered to be tolerant to this pressure at the benchmark.

Higher temperatures have been implicated in the proliferation of trematode parasites which have caused mass mortalities in the snail *Hydrobia ulvae* (Jensen & Mouritsen, 1992), which is often abundant in this biotope. No other information was found on tolerance of component species to increased temperature. Nevertheless, an increase in temperature may indirectly affect some species as microbial activity within the sediments will be stimulated increasing oxygen consumption and promoting hypoxia.

**Sensitivity assessment.** Typical surface water temperatures around the UK coast vary, seasonally from 4-19°C (Huthnance, 2010). It is likely that the important characteristic species are able to resist a long-term increase in temperature of 2°C and may resist a short-term increase of 5°C. *Limecola balthica* may retreat north as a result of long-term warming and climate change. However, the important characterizing species *Hediste diversicolor* are likely to survive a 5°C increase in temp for one month period, or 2°C for one year, although reproductive activities may be impacted. For instance, without colder winters spawning may not be synchronised and so recruitment would be reduced. A resistance of '**High**', a resilience of '**High**' (as longer lived later maturing species are present) and a sensitivity of '**Not sensitive'** have been assigned.

| | | | |
|---|---|---|---|
| **Temperature decrease (local)** | High<br>Q: High A: High C: High | High<br>Q: High A: High C: High | Not sensitive<br>Q: High A: High C: High |

A decrease in temperature has been shown to be 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* coincident with a reduction in the numbers of *Carcinus maenus* 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 (mean *Hediste diversicolor* density increased from 24/m$^2$ to 151/m$^2$ 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). A reduced rise in temperature is likely to limit this factor.

Colder winter temperatures have been shown to benefit *Limecola balthica* population dynamics. Recruitment success increased following colder winters and repeated recruitment failure has occurred after mild winters in comparable North Sea location (Beukema, 1992, Schueckel & Kroencke, 2013; Beukema *et al*., 2001). In Friedrichskoog, Germany, König (1943) found a high accumulation of dead *Cerastoderma edule* biomass after a severe winter 1936/1937 but high numbers of *Limecola balthica* (80,000 individuals/m$^2$) spat in following years (winter 1939). Winter water surface temperatures in the Wadden Sea (Netherlands) have increased 1.5°C since the 1980s (Oost *et al*., 2009). During milder winters greater body weight loss and production of fewer and smaller eggs has been observed in *Limecola balthica* (van der Meer *et al*., 2003). It is noted in the literature however, that reduced recruitment success during milder winters may also be due to increased predation as juvenile *Crangon crangon* have shown increased abundance in relation to milder winters (Beukema & Dekker, 2005).

The geographical distribution of *Limecola balthica* suggests that it is very tolerant of low temperature. The species occurs in the Gulfs of Finland and Bothnia where the sea freezes for several months of the year (Green, 1968). It must, therefore, resist much lower temperatures than it experiences in Britain and Ireland. Furthermore, *Limecola balthica* was apparently unaffected by the severe winter of 1962/3, which severely affected many other bivalve species (Crisp, 1964), and De Wilde (1975) noted that *Limecola balthica* kept at 0°C maintained a high level of feeding activity. It is likely, therefore, that in seas around the UK and Ireland *Limecola balthica* would be resist decreases in temperature at the pressure benchmark level.

*Eteone longa* recorded as arctic-boreal, both from the Atlantic and the Pacific and occurs throughout the North Sea. Occurring in the arctic-boreal region suggests the species is resistant to a decrease in temperature in UK and Irish seas.

Most littoral oligochaetes, including tubificids and enchytraeids, can survive freezing temperatures and can survive in frozen sediments (Giere & Pfannkuche, 1982). *Tubificoides benedii* (studied as *Peloscolex benedeni*) recovered after being frozen for several tides in a mudflat (Linke, 1939).

**Sensitivity assessment.** The important characterizing species show limited impacts and, potentially, benefits to abundance and recruitment from decreases in temperature. Therefore, a 5°C decrease in temp for one month period, or 2°C for one year is likely to have limited negative impact on all characterizing species in the biotope, within British and Irish seas. Hence, resistance is assessed as '**High**', resilience is assessed as '**High**', and sensitivity as **'Not Sensitive'**.

| | | | |
|---|---|---|---|
| **Salinity increase (local)** | Low | High | Not sensitive |
| | Q: Low A: NR C: NR | Q: High A: High C: Medium | Q: Low A: Low C: Low |

The biotopes occurs in variable (18-35 ppt), reduced (18-30 ppt) and full salinity (30-35 ppt) (JNCC, 2015). Biotopes occurring in variable and reduced salinity are not considered sensitive to a change to full salinity as this falls within the natural habitat range. However, species may be sensitive to an increase in the salinity regime to hypersaline (>40 ppt). Little evidence was found for tolerances of elevated salinity although populations are likely to be acclimated to short-term increased salinities in surface sediment layers as evaporation on hot days, or wind driven desiccation leads to increased salinities.

*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). Seitz (2011) found that the distribution of *Limecola balthica* (as *Macoma balthica*) across a salinity gradient between a minimum and maximum of 8.8psu to 19 psu in Cheaspeake Bay was not influenced by salinity. Instead, resource availability was the principal influence on *Limecola balthica*. McLusky & Allan (1976) reported that *Limecola balthica* (as *Macoma balthica*) failed to grow at 41 psu. It is likely that *Limecola balthica* would be tolerant of an increase in salinity category to fully marine but further increases to >40‰ are likely to affect growth and condition.

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.

*Pygospio elegans* is common in both marine and brackish waters in the Schelde estuary (Netherlands) suggesting in European habitats the species tolerates a broad salinity range (Ysebaert *et al.*, 1993). Studies of *Pygospio elegans* population structure in the Baltic Sea and North Sea also found larvae were not hampered by changes in salinity (Kesaniemi *et al.*, 2012). Although case studies are lacking for British and Irish coasts, the existing evidence suggests *Pygospio elegans* would tolerate salinity changes from variable to full but may be sensitive to a change from full salinity to hypersaline (>40 ppt).

Boyden & Russell (1972) stated that *Cerastoderma edule* prefers salinities between 15 and 35 psu. Russell & Peterson (1973) reported lower median salinity limits of 12.5 psu and upper median salinity limits of 38.5 psu. Rygg (1970) noted that *Cerastoderma edule* did not survive 23 days exposure to <10 psu or at 60 psu, although they did survive at 46 psu. Rygg (1970) also demonstrated that salinity tolerance was temperature dependent (after 3 days, 100% survival at 33 psu and 35-38°C, but 50% mortality occurred at 20 psu and 37°C and 100% mortality at 13 psu and 37°C). Wilson (1984) noted that *Cerastoderma edule* remained open during 1 hour exposure to salinities between 13.3 and 59.3 psu. It should be noted that the tolerances reported above depend on the duration of the experiment.

Kingston (1974) found that *Cerastoderma edule* larvae grew optimally at 30 and 35 psu, and grew well at 40 psu but the growth increment declined at 45 psu and larvae did not metamorphose. He noted that *Cerastoderma edule* larvae survived between 20 -50 psu, but died after 11 days at 55 psu or 10 days at 10 psu.

**Sensitivity assessment.** Little evidence was found to assess this pressure at the benchmark. Although species within the biotope are likely to tolerate short-term increases in salinity in sediment surface layers between tidal cycles, a longer change is likely to exceed salinity tolerances of adults and larvae. Biotope resistance is assessed as **'Low'** as some adults may survive and acclimate. Biotope resilience (following a return to suitable habitat conditions) is assessed as **'High'** and sensitivity is assessed as **'Low'.**

| | | | |
|---|---|---|---|
| **Salinity decrease (local)** | Low<br>Q: High A: Medium C: Medium | High<br>Q: High A: Low C: Medium | Low<br>Q: High A: Low C: Medium |

The biotope occurs in variable (18-35 ppt), redcued (18-30 ppt) and full salinity (30-35 ppt) (JNCC, 2015). Biotopes occuring in full salinity are considered tolerant of a change to variable or reduced salinity as this falls within the natural habitat range. The decrease in salinity assessed at the benchmark is to low salinity (<18 ppt). The available evidence (summarised below) suggests that the characterizing species are tolerant of a short-term decrease to low salinity but it is likely that for species such as *Limecola balthica* long-term reductions would lead to mortality.

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, have an adverse effect 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.

McLusky & Allan (1976) conducted salinity survival experiments with *Limecola balthica* (as *Macoma balthica*) over a period of 150 days. Survival times declined with decreased salinity. At 12 psu specimens survived 78 days, whilst specimens at 8.5 psu survived 40 days. Some specimens of *Limecola balthica* survived 2.5 days at 0.8 psu, which was apparently due to the animal's ability to clamp its valves shut in adverse conditions. McLusky & Allan (1976) also reported that *Limecola balthica* failed to grow (increase shell length) at 15 psu. *Limecola balthica* is found in brackish and fully saline waters (Clay, 1967b) so may tolerate a state of flux. Its distribution in combination with the experimental evidence of McLusky & Allan (1976) suggests that *Limecola balthica* is likely to be resistant to decreased salinity over a short period. A decline in salinity in the long-term may have implications for the species viability in terms of growth, and the distribution of the species may alter as specimens at the extremes retreat to more favourable conditions. Metabolic function should, however, quickly return to normal when salinity returns to original levels. Decreased salinity may also affect the ability of *Limecola balthica* to tolerate contaminants such as heavy metals (see Bryant *et al*., 1985 & 1985a). Usually, contaminants become more toxic at low salinity (Langston, W.J. pers comm.).

Muus (1967) revealed that *Hydrobia ulvae* did not crawl into water with a salinity lower than 9□⁄□□. Hylleberg (1975) also found that under controlled conditions of salinity ranging from 10 to 30□⁄□□ and temperatures ranging from 5 to 35°C, shows that *Hydrobia ulvae* has maximal egestion at the combination of high salinity (30□⁄□□) and high temperature (30° C).

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.

**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 but *Limecola balthica* is likely to reduce in low salinity conditions. A similar biotope could remain where salinities were close to 18 ppt but a severe reduction in salinity would probably lead to loss of the biotope. Resistance is therefore assessed as **'Low'**. Resilience (following restoration of typical conditions) is **'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<br>Q: Low A: NR C: NR | High<br>Q: High A: Low C: Medium | Low<br>Q: Low A: Low C: Low |

*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). Highest abundances occur in very weak (negligible) to weak < 1 knot (<0.5 m/sec.) currents. These conditions are provided by this biotope, which occurs in extremely sheltered gravelly mud to gravelly sandy mud on the mid and lower shore.

The type direction and speed of the currents control sediment deposition within an area. Finer sediment will fall to the substratum in weaker currents. An increase in water flow rate would entrain and maintain particles in suspension and erode the mud. As a result the scouring and consequent redistribution of components of the substratum would alter the extent of suitable habitat available to populations of *Hediste diversicolor* and other species in the biotope that prefer finer sediment. Recovery of *Hediste diversicolor* would be influenced by the length of time it would take for the potential habitat to return to a suitable state for recolonization by adult and juvenile specimens from adjacent habitats, and the establishment of a breeding population. Recolonization may take between one and three years, as populations differ in reaching maturity (Dales, 1950; Mettam *et al.*, 1982; Olive & Garwood, 1981), from the time that the habitat again becomes suited to the species.

Coarser sediments are likely to remain in areas of strongest flow velocity (where finer particles have been re-suspended). Species such as *Pygospio elegans* and other opportunist polychaetes that tolerate coarser particle size will possibly increase in abundance. *Limecola balthica* is likely to experience greater impact from increased water flow as the species thrives in low energy environments, such as the extremely sheltered areas that characterize the biotope (Tebble, 1976). Increased water flow rate is likely to influence the sediment characteristics in this biotope, primarily by re-suspending and preventing deposition of finer particles (Hiscock, 1983). This is likely to result in erosion of the preferred habitat, which may cause mortality of some portion of the population of *Limecola balthica*. Higher current velocity (18 cm/s 0.18 m/s) recorded in flume experiments conducted in the Isle of Sylt (North Sea) led to juvenile *Limecola balthica* being washed out of the sediment (Zuhlke & Reise, 1994). Green (1968) reported that, towards the mouth of an estuary where sediments became coarser and cleaner, *Limecola balthica* was replaced by another tellin species, *Tellina tenuis*.

**Sensitivity assessment.** *Limecola balthica* abundance may be reduced if juveniles are washed from the substratum. Loss of mud content in some areas is possible and could lead to replacement by another species but this is unlikely at the pressure benchmark levels. An increase in flow velocity may alter the muddy sand sediments, resistance has been assessed as '**Medium**', recoverability is assessed as '**High**' and sensitivity is, therefore '**Low**'.

| | | | |
|---|---|---|---|
| **Emergence regime changes** | High<br>Q: High A: Medium C: High | High<br>Q: High A: High C: High | Not sensitive<br>Q: High A: Medium C: High |

The biotope is found on the lower, mid and upper shore (JNCC, 2015) and changes in emergence are unlikely to affect the biotope where it remains within an intertidal habitat.

*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 back to damper substrata. Gogina *et al.* (2010) 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.* (2010).

*Limecola balthica* occurs in the upper regions of the intertidal (Tebble, 1976) and is, therefore, likely to be tolerant of prolonged emergence. It is a bivalve and can close tightly by contraction of the adductor muscle, storing moisture inside the shell. The silty sediments in which the species lives have a high water content and are therefore resistant to desiccation. Furthermore, *Limecola balthica* is mobile and able to relocate in the intertidal by burrowing (Bonsdorff, 1984) or floating (Sörlin, 1988). It would be expected to react to an increase in emergence by migrating down the shore to its preferred position. There may be an energetic cost to this migration but it is not expected that mortality would result. *Limecola balthica* should quickly recover from the energetic cost of relocation. *Limecola balthica* occurs in the intertidal and sublittorally down to depths of 190 m (Olafsson, 1986), although is more abundant intertidally. Hence, it would be expected to resist a decrease in emergence regime. However, a case study, predicting changes in biomass of *Limecola balthica* in the Humber estuary, UK (western North Sea) under expected sea level rise conditions displayed negative impacts. Coastal squeeze from sea level rise would produce steeper and more homogenous beach face profiles. *Limecola balthica* was predicted to be lower on steeper beach faces and biomass of *Limecola balthica* was predicted to decrease (Fujii & Raffaelli, 2008).

*Tubificoides benedii* is capable of penetrating the substratum to depths of 10 cm, shows a 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*., 2010). Highest abundances were predicted by Gogina *et al.* (2010) 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.** As intertidal species, the biotope and characterizing species are found at a range of shore heights and are considered relatively resistant to changes in emergence which 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 at the land-ward 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. Biotope resistance is, therefore, assessed as '**High',** recoverability is assessed as '**High'** (by default) and the biotope is considered to be **'Not sensitive'.**

| | | | |
|---|---|---|---|
| **Wave exposure changes (local)** | High<br>Q: Low A: NR C: NR | High<br>Q: High A: High C: High | Not sensitive<br>Q: Low A: Low 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.

Wave action stimulates *Limecola balthica* to start burrowing and individuals have been shown to continue burrowing for a longer period of time than in still water (Breum, 1970). Limited zoobenthic biomass was recorded in areas exposed to strong currents and wave action (Beukema, 2002), limiting food availability, however,impacts from this pressure at the benchmark levels may be low for this biotope, as the biotope is limited to sheltered or extremely sheltered locations. Increases in wave action may therefore remain within the limits of the species tolerance but factors such as sediment redistribution may alter the physical biotope.

**Sensitivity assessment.** Resistance to a change in nearshore significant wave height >3% but <5% of the two main characterizing species *Hediste diversicolor* and *Limecola balthica* 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'** and the biotope is considered to be '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)<br>Q: NR A: NR C: NR | Not assessed (NA)<br>Q: NR A: NR C: NR | Not assessed (NA)<br>Q: NR A: NR C: NR |

This pressure is **Not assessed** but evidence is presented where available. The following review discusses impacts at higher concentrations than the pressure benchmark.

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).

*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).

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 LC$_{50}$ Zinc values for *Hediste diversicolor* at four salinities 5, 10, 17.5 and 30 psu at 12°C. The 96 hour LC$_{50}$ 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).

*Scrobicularia plana* can detect copper at a concentration of 0·01 ppm. The initial response is valve closure resulting in a rapid drop in heart rate. In concentrations of 0·05, 0·01 and, to a lesser extent, 0·1 ppm copper added to seawater, the clams begin to interact with the polluted water after 2–3 h. In 0·5 ppm, the valves remain closed and the heart rate is maintained at a low level over the 6-h exposure period. Mortality increases with time in 0·5 ppm copper concentration, reaching 50% in 5–7 days. In 0·05 and 0·01 ppm, no mortality was recorded over this period (Akberali & Black, 1980).Two experimental 30 day long static toxicity tests on small spat (2 to 3 mm in length) of *Scrobicularia plana* were undertaken on the effects of copper on the survival and burying activity in sand, of juveniles. Results showed that:

1) exposure to up to 80 µg Cu 1.' did not result in increased mortalities with respect to the controls;

2) Cu concentrations at 20 µg $l^{-1}$and above also increased the burying time of juveniles by the end of the experiment; and

3) the no-observed-effect concentration (NOEC) for Cu was the lowest dose tested (i.e. 10 µg I-'). (Ruiz *et al*., 1994).

Alterations of the burrowing behaviour of *Scrobicularia plana*, were studied in individuals exposed to soluble copper. Animals were exposed for 4 d to concentrations ranging from 25 to 150 ▯ g Cu l▯ 1. At the end of exposure, the burrowing kinetics in clean sediment were determined after 1 and 2 days. Even at the lowest tested concentrations, copper caused hypoactivity in organisms belonging to both species studied. Metabolical or physiological disturbances could be the cause of these impairments. Concentrations affecting burrowing behaviour were below those responsible for mortality in these species (Bonnard *et al*., 2009).

Laboratory tests in clean water can be misleading as these do not reflect lowered toxicity in the marine environment due to the buffering effects of carbon and sulphide which render copper non-labile (not bioavailable) and the influence of water pH, hardness, temperature and salinity etc. Field surveys have found that *Scrobicularia plana* is present in the highly contaminated Fal Estuary where levels of copper and zinc are high (Bryan & Gibbs, 1983).

| | | | |
|---|---|---|---|
| **Hydrocarbon & PAH contamination** | Not Assessed (NA)<br>Q: NR A: NR C: NR | Not assessed (NA)<br>Q: NR A: NR C: NR | Not assessed (NA)<br>Q: NR A: NR C: NR |

This pressure is **Not assessed** but evidence is presented where available. The following review discusses impacts at higher concentrations than the pressure benchmark.

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.

Analysis of *Scrobicularia plana* collected in April 2008 from three estuaries along a pollution gradient (Goyen < Loire < Seine) found sub-lethal effects on clams including neurotoxicity and impairment of digestive enzyme activities (cellulase or amylase) in the Loire and Seine estuaries. The highest lactate dehydrogenase activity was registered in the Loire estuary, in parallel with enhanced levels of vanadium (a metal present in petroleum), likely as a consequence of a small oil spill that occurred one month before the sampling collection. The median size was significantly lower in clams exposed to direct (chemicals) or indirect (available food) effects in the most contaminated site. Burrowing behaviour was disturbed in clams from both of the Loire and Seine estuaries, a response probably due to physiological impairment rather than to avoidance of contaminated sediment. The activation of defence mechanisms towards metals (metallothionein) and other classes of contaminants (the biotransformation enzyme glutathione-S-transferase) do not ensure a total protection since a number of impairments were observed at the infra-organismal (AChE and digestive enzyme activities) and individual (burrowing behaviour) levels in relation to the degree of anthropogenic pressure. However, even in the most contaminated estuary (Seine), historical records do not show a consistent decrease of *Scrobicularia plana* populations (cited from Boldina-Cosqueric *et al*., 2010)

| | | | |
|---|---|---|---|
| **Synthetic compound contamination** | Not Assessed (NA)<br>Q: NR A: NR C: NR | Not assessed (NA)<br>Q: NR A: NR C: NR | Not assessed (NA)<br>Q: NR A: NR C: NR |

This pressure is **Not assessed** but evidence is presented where available. The following review discusses impacts at higher concentrations than the pressure benchmark.

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:

1. 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 has been shown to be 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.
2. 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.

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 rate of decay of the synthetic chemical within the marine environment), recolonization by adult and juvenile specimens from adjacent habitats, and the establishment of a breeding population. This may take between one and three years, as populations differ in reaching maturity (Dales, 1950; Mettam *et al*., 1982; Olive & Garwood, 1981), from the time that the habitat again becomes suited to the species.

| | | | |
|---|---|---|---|
| **Radionuclide contamination** | No evidence (NEv)<br>Q: NR A: NR C: NR | Not relevant (NR)<br>Q: NR A: NR C: NR | No evidence (NEv)<br>Q: NR A: NR C: NR |

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.

Hutchins *et al*. (1998) described the effect of temperature on bioaccumulation by *Limecola balthica* of radioactive americium, caesium and cobalt, but made no comment on the intolerance of the species. Further, direct assessments of impacts at the benchmark pressure on benthic communities, and this biotope in particular were not found.

**Sensitivity assessment. No evidence.** Insufficient evidence was available to complete and assessment.

| | | | |
|---|---|---|---|
| **Introduction of other substances** | Not Assessed (NA)<br>Q: NR A: NR C: NR | Not assessed (NA)<br>Q: NR A: NR C: NR | Not assessed (NA)<br>Q: NR A: NR C: NR |

This pressure is **Not assessed**.

| | | | |
|---|---|---|---|
| **De-oxygenation** | High<br>Q: High A: High C: High | High<br>Q: High A: High C: High | Not sensitive<br>Q: High A: High C: High |

The habitats which *Hediste diversicolor* inhabits tend to have lower oxygen levels than other sediments. *Hediste diversicolor* is resistant to moderate hypoxia (Diaz & Rosenberg, 1995). Vismann (1990) demonstrated 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) 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 4 days in laboratory conditions and combined hypoxia and increased sulphide (1 mmo1 1-') for 3 days (Gamenick *et al*., 1996). Post larvae *Hediste diversicolor* were the only life stage to show les tolerance to hypoxia, surviving for only 14 hr (Gamenick *et al*., 1996).

*Limecola balthica* appears to be relatively tolerant of de-oxygenation. Brafield & Newell (1961) frequently observed that, in conditions of oxygen deficiency (e.g. less than 1 mg $O_2$/l), *Limecola balthica* (as *Macoma balthica*) displayed survivability of low oxygen concentrations and shell growth continued (Jansson *et al*., 2015). Although, sub-lethal effects of hypoxia have been identified as individual *Limecola balthica* moved upwards to fully expose itself on the surface of the sand or buried at shallower depths, leaving them at greater risk of predation (Long *et al*., 2014). Specimens lay on their side with the foot and siphons retracted but with valves gaping slightly allowing the mantle edge to be brought into full contact with the more oxygenated surface water lying between sand ripples. In addition, *Limecola balthica* was observed under laboratory conditions to extend its siphons upwards out of the sand in to the overlying water when water was slowly deoxygenated with a stream of nitrogen. The lower the oxygen concentration became the further the siphons extended.

This behaviour, an initial increase in activity stimulated by oxygen deficiency, is of interest because the activity of lamellibranchs is generally inhibited by oxygen deficient conditions (Brafield & Newell, 1961). Dries & Theede (1974) reported the following LT50 values for *Limecola balthica* (as *Macoma balthica*) maintained in anoxic conditions: 50 - 70 days at 5°C, 30 days at 10°C, 25 days at 15°C and 11 days at 20°C. Theede (1984) reported that *Limecola balthica* to resist extreme oxygen deficiency was mainly due to anaerobic metabolism. *Macoma balthica* is therefore very tolerant of hypoxia, although it may react by reducing metabolic activity and predation risk may increase. Metabolic function should quickly return to normal when oxygen levels are resumed and so recovery is expected.

The characterizing oligochaetes and polychaetes within the biotope that display tolerance to hypoxia include *Tubificoides benedii* and *Capitella capitata*, while *Pygospio elegans* is highly sensitive to hypoxia (Gogina *et al*., 2010). Exposure to dissolved oxygen concentration of less than or equal to 2 mg/l for 1 week is likely to limit *Pygospio elegans* abundance, whilst having limited impact on *Tubificoides benedii* and *Capitella capitata* populations.

**Sensitivity assessment.** Resistance to exposure to dissolved oxygen concentration of less than or equal to 2 mg/l for 1 week is assessed as 'High' for the characterizing species *Hediste diversicolor* and *Limecola balthica.* It is important to consider that other species that are common or abundant in the biotope may be impacted by decreased dissolved oxygen, such as *Pygospio elegans* and decreases in abundance of these species are likely. As this biotope is found in intertidal habitats oxygen levels will be recharged during the tidal cycle lowering exposure to this pressure for *Pygospio elegans*. Based on the reported tolerances for anoxia and intertidal habitat, biotope resistance is assessed as **'High'** resilience is assessed as '**High**' (by default) and the biotope is considered to be **'Not sensitive'** at the benchmark level

| | | | |
|---|---|---|---|
| **Nutrient enrichment** | High<br>Q: High A: High C: High | High<br>Q: High A: High C: High | Not sensitive<br>Q: High A: High C: High |

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 microalagae 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 SE 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 data base 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 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 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<br>Q: High A: High C: High | High<br>Q: High A: High C: High | Not sensitive<br>Q: High A: High C: High |

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 display 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 an input at the pressure benchmark is considered unlikely to lead to gross pollution effects . A resilience of '**High**' is assigned (by default) and the biotope is assessed as '**Not sensitive**'.

## Physical Pressures

| | Resistance | Resilience | Sensitivity |
|---|---|---|---|
| **Physical loss (to land or freshwater habitat)** | None<br>Q: High A: High C: High | Very Low<br>Q: High A: High C: High | High<br>Q: High A: High C: High |

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

| | | | |
|---|---|---|---|
| **Physical change (to another seabed type)** | None<br>Q: High A: High C: High | Very Low<br>Q: High A: High C: High | High<br>Q: High A: High C: High |

This biotope and sub-biotopes is only found in sediment, in particular, gravelly sandy mud or gravelly mud (JNCC, 2015). The burrowing organisms characterizing this biotope, including *Hediste diversicolor*, and *Limecola balthica* 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<br>Q: High A: Medium C: High | Very Low<br>Q: High A: High C: High | High<br>Q: High A: Medium C: High |

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 mud and sandy mud a change to finer sediments is not considered relevant as this falls within the natural habitat change (JNCC, 2015).

An increase in gravel and a change to clean sands or coarse sediments is likely to have a more significant effect as sediment cohesion and ability to retain organic matter and water is reduced altering habitat suitability for burrowing polychaetes and amphipods and deposit feeders.

*Hediste diversicolor* is infaunal and is reliant upon a muddy/sandy sediment in which to burrow. *Hediste diversicolor* has been identified in other intertidal sediments including gravels, clays and even turf (Clay, 1967; Scaps, 2002), although abundance may be reduced in these habitats. *Limecola balthica* is likely to tolerate increased gravel content as sediment was not shown to affect burrowing (Tallqvist, 2001), however, growth, shell size and body mass were greatest in higher sand content sediment and lower in higher gravel content sediments (Azouzi *et al.* 2002), suggesting long-term health and abundance may be affected by a permanent increase in gravel content.

Coarser sediments provide inhospitable conditions for colonizing infauna, although *Pygospio elegans* show greater tolerance of increased gravel content and are viewed as opportunistic species that are capable of exploiting these inhospitable conditions (Gray, 1981). Therefore, *Pygospio elegans* are likely to be less affected and even increase in abundance under a change in Folk class to gravelly mud (or a change from sandy mud to muddy sand, or gravelly muddy sand). *Capitella capitata* are likely to decrease in abundance as Degraer *et al.* (2006) found that *Capitella capitata* in the Belgium part of the North Sea were almost completely absent in sediments without mud. Similar species that prefer higher organic content may also show limited abundance in more gravelly sediments.

**Sensitivity assessment.** Case studies display decreasing abundance with increased gravel content (*Hediste diversicolor*) and reduced growth rates (*Limecola balthica*). Abundance of abundant polychaetes is likely to depend on each species tolerance of increasing gravel content, with species that can exploit the conditions increasing in abundance (*Pygospio elegans)* but other species decreasing in abundance. Resistance to a change in one Folk class is assessed as **'Low'** as increased gravel content is likely to lead to reduced abundance of characterizing species and result in biotope reclassification to the mixed sediment biotope LS.LMx.GvMu . Resilience is assessed as **'Very Low**' as a change at the benchmark is permanent. The sensitivity of the biotope overall is, therefore, considered to be '**High**'.

| | | | |
|---|---|---|---|
| **Habitat structure changes - removal of substratum (extraction)** | None<br>Q: High A: High C: High | High<br>Q: High A: High C: Medium | Medium<br>Q: High A: High C: Medium |

The substratum of this biotope consists of gravelly sandy mud or gravelly mud (Conner *et al.*, 2004). The characterizing infaunal species, including *Hediste diversicolor*, *Eteone longa* and *Limecola balthica* 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 extraction of 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 following 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<br>Q: High A: High C: Medium | High<br>Q: High A: High C: Medium | Low<br>Q: High A: Medium C: Medium |

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, although siphons of bivalves and tubes of the sedentary polychaete *Pygospio elegans*, may project above the sediment surface. Damage to tubes and siphons would require repair. 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).

**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 bivalves, tube dwelling polychaetes and *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<br>Q: High A: High C: Medium | High<br>Q: High A: High C: Medium | Low<br>Q: High A: High C: Medium |

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 shells of *Limecola balthica* provide some protection. *Pygospio elegans* inhabits fragile tubes at the sediment surface and *Hydrobia ulvae* crawl on the sediment, both species are likely to be vulnerable to penetration and disturbance of the sediment.

The effects of a 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 recolonisation 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).

Ferns *et al*. (2000) studied effects of harvesting of cockles and reported a decline in muddy sands of 83% in *Pygospio elegans* (initial density 1850 $m^{-2}$) when a mechanical tractor towed harvester was used in a cockle fishery. *Pygospio elegans* and *Hydrobia ulvae* were significantly depleted for >100 days after harvesting (surpassing the study monitoring timeline).

Rees, 1978, (cited in Hiscock *et al*., 2002), assessed pipe laying activities. The pipe was laid in a trench dug by excavators and the spoil from the trenching was then used to bury the pipe. The trenching severely disturbed a narrow zone, but a zone some 50 m wide on each side of the pipeline was also disturbed by the passage of vehicles. The tracked vehicles damaged and exposed shallow-burrowing species such as the bivalves *Cerastoderma edule* and *Limecola balthica*, which were then preyed upon by birds. During the construction period, the disturbed zone was continually re-populated by mobile organisms, such as *Hydrobia ulvae*.

**Sensitivity assessment.** 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<br>Q: Low A: NR C: NR | High<br>Q: High A: High C: High | Not sensitive<br>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.** The following sensitivity assessment relies on expert judgement, utilising evidence of species traits and distribution and therefore confidence has been assessed as low. 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. Sensitivity of the biotope is, therefore, assessed as '**Not Sensitive**'.

| | | | |
|---|---|---|---|
| **Smothering and siltation rate changes (light)** | Medium<br>Q: High A: High C: Medium | High<br>Q: High A: High C: High | Low<br>Q: High A: High C: Medium |

The degree to which the characterizing species are able to 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).

*Limecola balthica* is able to burrow both vertically and horizontally through the substratum. It is likely that *Limecola balthica* is not sensitive to smothering by a layer of sediment 5 cm thick as it is a mobile species able to burrow upwards and surface from a depth of 5 - 6 cm (Brafield & Newell, 1961; Brafield, 1963; Stekoll *et al*., 1980). Turk and Risk (1981) investigated the effect of experimentally induced sedimentation (through fences and boxes that induced sediment deposition on intertidal mudflats in the Bay of Fundy), of 1-3.5cm at a rate of 1.9-10.2 cm/month. The results showed that *Limecola balthica* was generally unaffected.

The associated species *Pygospio elegans* is limited by high sedimentation rates (Nugues *et al*., 1996) and the species does not appear to be well adapted to oyster culture areas where there are high rates of accumulation of faeces and pseudo faeces (Sornin *et al*., 1983; Deslous-Paoli *et al*., 1992; Mitchell, 2006 and Bouchet & Sauriau 2008). *Pygospio elegans* is known to decline in areas following re-deposition of very fine particulate matter (Rhoads & Young, 1971; Brenchley, 1981).

Experimental relaying of mussels on intertidal fine sands led to the absence of *Pygospio elegans* compared to adjacent control plots. The increase in fine sediment fraction from increased sediment deposition and biodeposition alongside possible organic enrichment and decline in sediment oxygen levels was thought to account for this (Ragnarsson & Rafaelli, 1999).

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 '**Medium**' as some species associated with the biotope such as *Corophium volutator* and *Pygospio elegans* may decline but the biotope is likely to be recognizable within a week due to repositoing and migration of mobile species. Resilience is assessed as '**High**' and sensitivity is assessed as '**Low**.

| | | | |
|---|---|---|---|
| **Smothering and siltation rate changes (heavy)** | Low<br>Q: High A: High C: Medium | High<br>Q: High A: Low C: Medium | Low<br>Q: High A: Low C: Medium |

Studies have found that beach 'replenishment' or 'nourishment' that involves the addition of sediments on beaches can have a number of impacts on the infauna (Peterson *et al*., 2000; Peterson *et al*., 2006). Impacts are more severe when the sediment added differs significantly in grain size or organic content 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 which 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.

Witt *et al.* (2004) identified an increase in *Limecola balthica* (as *Macoma balthica*) abundance in areas of disposal of dredge waste spoil, possibly due to nutrient input at the disposal site. This suggests *Limecola balthica* responds opportunistically to this pressure.

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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.** 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, but 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)<br>Q: NR A: NR C: NR | Not assessed (NA)<br>Q: NR A: NR C: NR | Not assessed (NA)<br>Q: NR A: NR C: NR |

Examples of the impact of specific marine litter, including cigarette butts and micro-plastics are also considered..

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). This UK study suggests health of infauna populations are negatively impacted by this pressure.

Studies of other characterizing species in relation to micro plastics were not available. However, studies of sediment dwelling, sub surface deposit feeding worms, showed negative impacts from ingestion of micro plastics. 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 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 colonisation 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 33 m□ of micro-plastics a year.

**Sensitivity assessment.** Marine litter in the form of cigarette butts or micro plastics health of populations of characterizing species may be impacted. Significant impacts have been shown in laboratory studies but impacts at biotope scales are still unknown. Evidence and confidence in the assessment is limited and this pressure is **'Not assessed'**.

| | | | |
|---|---|---|---|
| **Electromagnetic changes** | No evidence (NEv)<br>Q: NR A: NR C: NR | No evidence (NEv)<br>Q: NR A: NR C: NR | No evidence (NEv)<br>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 behaviour of predators and affect infauna populations. Evidence is limited and occurs for electric and magnetic fields below the benchmark levels, confidence in evidence of these effects is very low.

Field measurements of electric fields at North Hoyle wind farm, North Wales recorded 110µ V/m (Gill *et al.*, 2009). Modelled results of magnetic fields from typical subsea electrical cables, such as those used in the renewable energy industry produced magnetic fields of between 7.85 and 20 µT (Gill *et al.* 2009; Normandeau *et al.* 2012). Electric and magnetic fields smaller than those recorded by in field measurements or modelled results were shown to create increased movement in potential predators of *Hediste diversicolor*, 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)<br>Q: NR A: NR C: NR | Not relevant (NR)<br>Q: NR A: NR C: NR | Not relevant (NR)<br>Q: NR A: NR C: NR |

Species within the biotope can probably detect vibrations caused by noise and in response may retreat in to the sediment for protection. However, at the benchmark level the community is unlikely to be sensitive to noise and this therefore is '**Not relevant'.**

| | | | |
|---|---|---|---|
| **Introduction of light or shading** | Not relevant (NR)<br>Q: NR A: NR C: NR | Not relevant (NR)<br>Q: NR A: NR C: NR | Not relevant (NR)<br>Q: NR A: NR C: NR |

There is no direct evidence of 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 limited direct impact is expected. As this biotope is not characterized by the presence of primary producers, shading is unlikely to alter the character of the habitat directly and this pressure is considered **'Not relevant'**.

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. 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<br>Q: Low A: NR C: NR | High<br>Q: High A: High C: High | Not sensitive<br>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). Evidence on this pressure is limited.

The trait of *Hediste diversicolor* to lay and protect eggs within a burrow is likely to limit the impact of barriers to movement on populations. The ability of postlarvae, larger juveniles, and adults of *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. A barrier to movement is likely to limit colonization outside the enclosed area, but increase populations within the enclosed area

*Capitella capitata* and the associated species *Pygospio elegans* are capable of both benthic and pelagic dispersal. In the sheltered waters where this biotope occurs, with reduced water exchange, in-situ reproduction may maintain populations rather than long-range pelagic dispersal. As the tubificid oligochaetes that characterize this biotope have benthic dispersal strategies (via egg cocoons laid on the surface (Giere & Pfannkuche, 1982), water transport is not a key method of dispersal over wide distances.

**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)<br>Q: NR A: NR C: NR | Not relevant (NR)<br>Q: NR A: NR C: NR | Not relevant (NR)<br>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

| | | | |
|---|---|---|---|
| **Visual disturbance** | Not relevant (NR)<br>Q: NR A: NR C: NR | Not relevant (NR)<br>Q: NR A: NR C: NR | Not relevant (NR)<br>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)<br>Q: NR A: NR C: NR | Not relevant (NR)<br>Q: NR A: NR C: NR | Not relevant (NR)<br>Q: NR A: NR C: NR |

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

| | | | |
|---|---|---|---|
| **Introduction or spread of invasive non-indigenous species** | Low<br>Q: High A: High C: High | Very Low<br>Q: Low A: NR C: NR | High<br>Q: Low A: Low C: Low |

Intertidal 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, 2009b).

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.* 2004, 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<br>Q: Low A: NR C: NR | High<br>Q: High A: High C: High | Not sensitive<br>Q: Low A: Low C: Low |

Limited evidence was returned by searches on the effect on *Hediste diversicolor* of 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 that can cause large scale mortality in shrimp in *Hediste diversicolor* by both feeding and immersion.

A recent review of parasites, pathogens and commensals identified a range of agents impacting European cockles, including 50 conditions affecting *Cerastoderma edule* (Longshaw & Malham, 2013). Cockles are hosts to viruses, bacteria, fungi, Apicomplexa , Amoeba, Ciliophora, Perkinsozoa, Haplosporidia, Cercozoa, Turbellaria, Digenea, Nematoda, Crustacea and Nemertea. Mortalities are associated particularly with digeneans and some protistan infections; parasites may limit growth, reduce fecundity and alter burrowing behaviour (Longshaw & Malham, 2013). Parasites and disease are more likely to cause mortalities in populations that are subject to suboptimal conditions or other stressors such as hot summers or cold winters (Longshaw & Malham, 2013). Experimental infection of Cerastoderma edule with a trematode parasite showed that effects differed depending on habitat conditions (Wegeberg & Jensen, 2003). Infected *Cerastoderma edule* reared in sub-optimal conditions lost more body weight than infected cockles in more optimal habitats and did not regain condition when placed in higher shore habitats where immersion and food supply was limited. Infected cockles placed on lower shore sites with longer emersion times regained condition despite the infection and were equivalent to controls. The impact of trematodes is therefore mediated by habitat conditions and in some instances may have no effect (Wegeberg & Jensen, 2003).

**Sensitivity assessment.** Biotope resistance is assessed as '**High' a**s mass mortalities of key characterizing species have not been reported, resilience is assessed as '**High**' and sensitivity is, therefore assessed as '**Not sensitive**'.

| | | | |
|---|---|---|---|
| **Removal of target species** | Low<br>Q: High A: Medium C: Medium | High<br>Q: High A: Medium C: Medium | Low<br>Q: High A: Medium C: High |

Ragworms *Hediste diversicolor* are harvested as bait for angling while the abundant bivalves in the biotope *Cerastoderma edule* and *Mya arenia* are harvested commercially and recreationally. Removal of target species such as cockles *Cerastoderma edule* or bait digging for *Hediste diversicolor* is likely to impact the biotope. The extent of the impact will depend on the fishing / removal method and spatial extent.

Hall & Harding (1997) examined the effects of hydraulic and tractor dredging of *Cerastoderma edule* on macrobenthic communities. They concluded that although significant mortality of *Cerastoderma edule* and other infauna occurred, recovery was rapid and the overall effects on populations was low. Hall & Harding (1997) found that *Cerastoderma edule* abundance had returned to control levels within about 56 days and Moore (1991) also suggested that recovery was rapid. Cotter *et al.* (1997) noted that tractor dredging reduced the *Cerastoderma edule* stock by 31-49% depending on initial density, while Pickett (1973) reported that hydraulic dredging removed about one third of the cockle fishery. Tractor dredging leaves visible tracks in the sediment, which can act as lines for erosion and accelerate erosion of the sediment (Moore, 1991; Gubbay & Knapman, 1999). In most cases subsequent settlement was good especially in areas of previously high population density, however, Franklin & Pickett (1978) noted that subsequent spat survival was markedly reduced. Cotter et al. (1997) reported appreciable loss of spat and juveniles, partly due increased predation of exposed juveniles. Pickett (1973) also noted reduced survivability of 1-2 year old cockles after hydraulic dredging. However, most studies concluded that the impact of mechanised dredging on cockle populations and macrofauna in the long term was low (Pickett, 1973; Franklin & Pickett, 1978; Cook, 1990; Moore, 1991; Cotter et al., 1997; Hall & Harding, 1997; Ferns et al., 2000). Time of year of exploitation will influence recovery and avoiding seasonal spawning or larval settlement periods is likely to reduce the time taken for recovery (Gubbay & Knapman, 1999). Cockle beds have been mechanically fished for decades but several beds are closed from time to time depending on settlement and recruitment to the population, which is sporadic. Recovery may take less than a year in years of good recruitment but longer in bad years.

Following experimental removal of large adult *Cerastoderma edule* by Frid & Casear (2012) sediments showed increased biodiversity and assemblages dominated by traits common to opportunist taxa at a species-poor shore at Warton Sands, Morecambe Bay, and a more diverse shore at Thurstaston, Dee estuary. The movements of cockles disturb and exclude the amphipod *Corophium volutator* and other species (Flach, 1996; Flach & de Bruin, 1994) the removal of cockles may, therefore, allow this species to colonize intertidal flats. During periods of low cockle density, Desprez *et al*., (1992) observed that *Pygospio elegans* established dense populations; when cockles returned these were lost within one year.

**Sensitivity assessment.** Removal at a commercial or recreational scale is assessed as not affecting the entire extent of the biotope, but affecting patches within the biotope. Due to potential impacts on *Hediste diversicolor* populations, in particular females, and impacts on *Cerastoderma edule* populations the biotope is likely to be sensitive to this pressure. The abundance of other soft-sediment infauna (particularly opportunist species such as *Pygospio elegans* and *Capitella capitata* may increase in disturbed patches in the short-term as a result of the removal of cockles resulting in reduced competition for space and predation on larvae). Where sediments remain suitable cockles are likely to recolonize via adult migration, survival of small, discarded cockles or via larval recruitment.  In general fishing practices will be efficient at removing *Hediste diversicolor* and *Cerastoderma edule.* Resistance is therefore assessed as '**Low**' (removal is not considered to be total as smaller individuals are not retained and harvesting is unlikely to be 100% efficient at removing larger cockles). Resilience is assessed as '**High**' (although *Cerastoderma edule* many not recover within this timescale due to episodic recruitment; see *Cerastoderma edule* dominated biotopes). Biotope sensitivity is assessed as '**Low**'.

| | | | |
|---|---|---|---|
| **Removal of non-target species** | Low<br>Q: Low A: NR C: NR | High<br>Q: High A: Medium C: Medium | Low<br>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 would alter the physical structure of the habitat resulting in the loss of the ecosystem functions such as secondary production performed by these species.

McLusky *et al.* (1983) found that *Limecola balthica* populations were unaffected in dug areas, following bait digging for lugworms suggesting the biotope would recover from this impact if it occurred over a limited spatial scale.  However, Hiddink (2003) shows density of *Limecola balthica* was reduced in areas in the Wadden sea (Netherlands) that had experienced suction dredging for cockles, which removes the surface sediment.

Incidental removal of the characterizing species would alter the character of the biotope and the delivery of ecosystem services such as secondary production and bioturbation. Populations of characterizing species, such as *Hediste diversicolor* and *Limecola balthica* provide food for macroinvertebrates fish and birds.

The removal of predators such as shrimp and crab may enhance recruitment of larvae of characterizing species (Beukema & Dekker, 2005). The physical effects of removal of other species such as polychaete worms targeted by bait diggers may, however, impact *Eteone longa*, *Cerastoderma edule* and other species associated with this biotope, through direct damage and removal and displacement. These direct effects of sediment disturbance are assessed in the physical damage sections.

The removal of *Hediste diversicolor*  and *Cerastoderma edule* and other associated species would alter the biotope from the description and change community structure (diversity, biomass and abundance), potentially altering ecosystem function and the delivery of ecosystem goods and services (including the supply of food to fish and birds).

**Sensitivity assessment.** The assessment considers whether the removal of characterizing and associated species as by-catch would impact the biotope. Lethal damage to and removal of *Hediste diversicolor* and *Cerastoderma edule* and other species as by-catch would alter the character of the biotope. As *Cerastoderma edule* and *Limecola balthica* and other abundant bivalve species are either sedentary or incapable of rapid evasive movements, resistance is assessed as '**Low**'. Resilience is assessed as '**High**'.  *Cerastoderma edule* may not be recovered in this timescale due to episodic recruitment but this is not considered likely to alter biotope classification(see *Cerastoderma edule* dominated biotopes) and sensitivity is therefore categorized as '**Low**'.  Physical damage to the sediment and other physical damage factors are considered in the abrasion and extraction pressures.

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