



# MarLIN

## Marine Information Network

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

# Semi-permanent tube-building amphipods and polychaetes in sublittoral sand

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

Eliane De-Bastos & Will Rayment

2016-07-07

A report from:

The Marine Life Information Network, Marine Biological Association of the United Kingdom.

**Please note.** This MarESA report is a dated version of the online review. Please refer to the website for the most up-to-date version [<https://www.marlin.ac.uk/habitats/detail/136>]. All terms and the MarESA methodology are outlined on the website (<https://www.marlin.ac.uk>)

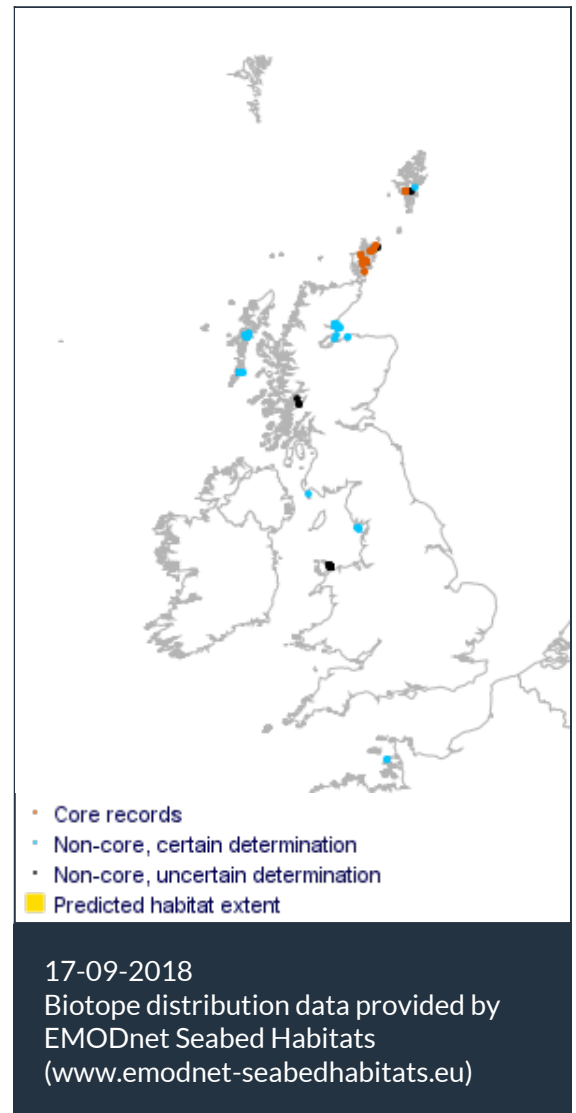
This review can be cited as:

De-Bastos, E. & Rayment, W.J. 2016. Semi-permanent tube-building amphipods and polychaetes in sublittoral sand. 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.136.1>



The information (TEXT ONLY) provided by the Marine Life Information Network (MarLIN) is licensed under a Creative Commons Attribution-Non-Commercial-Share Alike 2.0 UK: England & Wales License. Note that images and other media featured on this page are each governed by their own terms and conditions and they may or may not be available for reuse. Permissions beyond the scope of this license are available [here](#). Based on a work at [www.marlin.ac.uk](http://www.marlin.ac.uk)

(page left blank)



Researched by Eliane De-Bastos & Will Rayment

Referred by Admin

## Summary

### ☰ UK and Ireland classification

<b>EUNIS 2008</b>	A5.234	Semi-permanent tube-building amphipods and polychaetes in sublittoral sand
<b>JNCC 2015</b>	SS.SSa.IFiSa.TbAmPo	Semi-permanent tube-building amphipods and polychaetes in sublittoral sand
<b>JNCC 2004</b>	SS.SSa.IFiSa.TbAmPo	Semi-permanent tube-building amphipods and polychaetes in sublittoral sand
<b>1997 Biotope</b>	SS.IMU.MarMu.TubeAP	Semi-permanent tube-building amphipods and polychaetes in sublittoral mud or muddy sand

### 🔍 Description

Sublittoral marine sand in moderately exposed or sheltered inlets and voes in shallow water may support large populations of semi-permanent tube-building amphipods and polychaetes. Typically

dominated by *Crassikorophium crassicorne* (syn. *Corophium crassicorne*) with other tube building amphipods such as *Ampelisca* spp. also common. Other taxa include typical shallow sand fauna such as *Spiophanes bombyx*, *Urothoe elegans*, *Bathyporeia* spp. along with various polychaetes including *Parexogone hebes* (syn. *Exogone hebes*) and *Lanice conchilega*. *Polydora ciliata* may also be abundant in some areas. At the sediment surface, *Arenicola marina* worm casts may be visible and occasional seaweeds such as *Saccharina latissima* (syn. *Laminaria saccharina*) may be present. As many of the sites featuring this biotope are situated near to fish farms it is possible that it may have developed as the result of moderate nutrient enrichment. The distribution of this biotope is poorly known and like the muddier SS.SMu.ISaMu.AmpPlon, to which it is related, appears to have a patchy distribution (Information taken from Connor *et al.*, 2004).

### ↓ Depth range

-

### Additional information

None

### ✓ Listed By

- none -

### Further information sources

Search on:



## Habitat review

### 🔄 Ecology

#### Ecological and functional relationships

- The biotope is characterized by tube-building polychaetes and amphipods, with errant polychaetes and nemerteans foraging in the surrounding and underlying sediment.
- The dominant tube-builders are the deposit feeding polychaetes *Polydora ciliata*, *Spiophanes bombyx* and *Pygospio elegans*. In areas of mud, the tubes built by *Polydora ciliata* can agglomerate and form layers of mud an average of 20 cm thick, occasionally up to 50 cm (Daro & Polk, 1973). The tube-building, suspension feeding amphipods *Ampelisca* sp. are present where the biotope occurs in shallow warm waters, while they are probably replaced by the very similar *Haploops tubicola* in deeper, colder waters (Dauvin & Bellan-Santini, 1990).
- The feeding activities of high densities of *Polydora ciliata* may inhibit the establishment of other benthic species by removing settling and developing larvae (Daro & Polk, 1973).
- Infaunal deposit feeding polychaetes include the burrow dwelling *Arenicola marina*, the sedentary *Chaetozone setosa*, the mobile detritivore *Scoloplos armiger* and species tolerant of nutrient enrichment including *Capitomastus minimus* and *Capitella* sp.
- The amphipods and the infaunal annelid species in the biotope probably interfere strongly with each other. Adult worms probably reduce amphipod numbers by disturbing their burrows and tubes, while high densities of amphipods can prevent establishment of worms by consuming larvae and juveniles (Olafsson & Persson, 1986).
- The biotope contains a number of infaunal bivalve species, including *Abra alba*, *Fabulina fabula* and *Kurtiella bidentata*, which probably both deposit feed and suspension feed depending on local environmental conditions.
- Spatial competition probably occurs between the infaunal suspension feeders and deposit feeders. Reworking of sediment by deposit feeders, e.g. *Arenicola marina*, makes the substratum less stable, increases the suspended sediment and makes the environment less suitable for suspension feeders (Rhoads & Young, 1970). Tube building by amphipods stabilizes the sediment and arrests the shift towards a community consisting entirely of deposit feeders.
- Amphipods are predated chiefly by nemertean worms. For example, the nemertean *Nipponnemertes pulcher* is the dominant predator in the *Haploops* community in the Danish Oeresund (McDermott, 1984).
- Mobile, carnivorous polychaetes, including *Anaitides mucosa*, *Eteone longa*, *Nephtys hombergi* and *Pholoe inornata*, predate the smaller annelids and crustaceans.

#### Seasonal and longer term change

Temporal changes are likely to occur in the community due to seasonal recruitment processes. For example, the early reproductive period of *Polydora ciliata* often enables the species to be the first to colonize available substrata (Green, 1983). The settling of the first generation in April is followed by the accumulation and active fixing of mud continuously up to a peak during the month of May. The following generations do not produce a heavy settlement due to interspecific competition and heavy mortality of the larvae (Daro & Polk, 1973). Later in the year, the surface layer cannot hold the lower layers of the mud mat in place and they may be swept away by water currents. The substratum may now be colonized by the abundant larvae of other species in the water column.

There is a seasonal variation in planktonic production in surface waters which probably affects the food supply of the benthos in the biotope. Increased production by phytoplankton in spring and summer due to increased temperatures and irradiance is followed by phytoplankton sedimentation events which are correlated with seasonal variations in the organic content of benthic sediments (Thouzeau *et al.*, 1996). These variations directly influence the food supply of the deposit feeders and suspension feeders in the biotope.

Where the biotope occurs in the shallow subtidal, it is likely to be affected by winter storms. Storms may cause dramatic changes in distribution of macro-infauna by washing out dominant species, opening the sediment to recolonization by adults and/or available spat/larvae (Eagle, 1975; Rees *et al.*, 1976; Hall, 1994) and by reducing success of recruitment by newly settled spat or larvae (see Hall, 1994 for review). For example, during winter gales along the North Wales coast large numbers of *Abra alba* and *Kurtiella bidentata* were cast ashore and over winter survival rate was as low as 7% and 50% respectively in the more exposed locations (Rees *et al.*, 1976). Sediment transport and the risk of smothering also occurs.

### Habitat structure and complexity

- Structural complexity is provided by the many tube building species in the biotope. The principal tube builders are the polychaetes *Polydora ciliata* and *Spiophanes bombyx* and the amphipods *Ampelisca* sp. and *Haploops tubicola*. The tubes built by *Polydora ciliata* for example are embedded in the sediment and the ends extend a few millimetres above the substratum surface. The mats of agglomerated sediment may be up to 50 cm thick.
- High densities of tube builders and the presence of tubes favours further sedimentation of fine particles (e.g. Mills (1967) for *Ampelisca vadorum* and *Ampelisca abdita*) and may be a factor in stimulating recruitment of species such as *Haploops tubicola* (Glemarec *et al.*, 1986, cited in Dauvin & Bellan-Santini, 1990).
- Additional structural complexity is provided by the burrows of infauna although these are generally simple. 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 of species such as *Arenicola marina* 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). Underlying sediments may also become oxygenated by the activities of amphipods within their tubes (Mills, 1967).

### Productivity

Production in IMU.TubeAP is mostly secondary, derived from detritus and organic material. Where, the biotope occurs in shallow subtidal waters, some primary production comes from benthic microalgae (microphytobenthos e.g. diatoms, flagellates and euglenoides) and water column phytoplankton. Beyond 30m depth, there is unlikely to be any *in situ* primary production. In all cases, the benthos is supported predominantly by pelagic production and by detrital materials emanating from the coastal fringe (Barnes & Hughes, 1992). The amount of planktonic food reaching the benthos is related to:

- depth of water through which the material must travel;
- magnitude of pelagic production;
- proximity of additional sources of detritus;
- extent of water movement near the sea bed, bringing about the renewal of suspended supplies (Barnes & Hughes, 1992).

Food becomes available to deposit feeders by sedimentation on the substratum surface and by translocation from the water column to the substratum through production of pseudofaeces by suspension feeders.

Productivity in the biotope is expected to be high. The amphipods in particular have a short lifespan, grow to maturity quickly and have multiple generations per year.

The sediment in the biotope may be nutrient enriched due to proximity to anthropogenic nutrient sources such as sewage outfalls or eutrophicated rivers.

### Recruitment processes

- The spawning period for *Polydora ciliata* in northern England is from February until June and three or four generations succeed one another during the spawning period (Gudmundsson, 1985). After a week, the larvae emerge and are believed to have a pelagic life from two to six weeks before settling (Fish & Fish, 1996). The larvae settle preferentially on substrates covered with mud (Lagadeuc, 1991).
- The mating system of amphipods is polygynous and several broods of offspring are produced, each potentially fertilised by a different male. There is no larval stage and embryos are brooded in a marsupium, beneath the thorax. Embryos are released as subjuveniles with incompletely developed eighth thoracopods and certain differences in body proportions and pigmentation. Dispersal is limited to local movements of these subjuveniles and migration of the adults and hence recruitment is limited by the presence of local, unperturbed source populations (Poggiale & Dauvin, 2001). Dispersal of subjuveniles may be enhanced by the brooding females leaving their tubes and swimming to uncolonized areas of substratum before the eggs hatch (Mills, 1967).
- The tube building polychaetes, e.g. *Pygospio elegans*, generally disperse via a pelagic larval stage (Fish & Fish, 1996) and therefore recruitment may occur from distant populations. However, dispersal of the infaunal deposit feeders, such as *Scoloplos armiger* and *Arenicola marina*, occurs through burrowing of the benthic larvae and adults (Beukema & de Vlas, 1979; Fish & Fish, 1996). Recruitment must therefore occur from local populations or by longer distance dispersal during periods of bedload transport. Recruitment is therefore likely to be predictable if local populations exist but patchy and sporadic otherwise.

### Time for community to reach maturity

A community containing *Polydora ciliata* is likely to reach maturity very rapidly because *Polydora ciliata* is a short lived species that reaches maturity within a few months and has three or four spawnings during a breeding season. For example, in colonization experiments in Helgoland (Harms & Anger, 1983) *Polydora ciliata* settled on panels within one month in the spring. The tubes built by *Polydora ciliata* agglomerate sometimes to form layers of mud up to 20cm thick. However, it may take several years for a *Polydora ciliata* 'mat' to reach a significant size.

The life cycles of amphipods varies between the different families. Based on the intertidal species, *Corophium volutator*, the *Corophium* sp. may produce several broods over the summer breeding season (Fish & Fish, 1996). *Haploops tubicola* produces 1 or 2 broods per year with a longevity of 2 or 3 years (Dauvin & Bellan-Santini, 1990) and Mills (1967) reported that *Ampelisca vadorum* and *Ampelisca abdita* produced only 1 brood per generation but there were 2 or more generations per year. In the English Channel, two reproductive patterns were identified. Species such as *Ampelisca tenuicornis* and *Ampelisca typica* produced two generations per year. The juveniles born in May-June were able to brood in September-October (Dauvin, 1988b; Dauvin, 1988c). Species such as *Ampelisca armoricana* and *Ampelisca sarsi* produced only one brood per generation and per year (Dauvin, 1989; Dauvin, 1988d). *Ampelisca brevicornis* showed an intermediate cycle with one

generation per year during cold years (cold spring) and two generations per year during warm years (warm spring) and its cycle is intermediate between univoltine cycle and bivoltine cycle (Dauvin, 1988b,c,d,e; Dauvin, 1989, Dauvin & Bellan-Santini, 1990).

### Additional information

-

## Preferences & Distribution

### Habitat preferences

#### Depth Range

#### [Water clarity preferences](#)

**Limiting Nutrients** No information found

#### Salinity preferences

#### Physiographic preferences

#### Biological zone preferences

#### Substratum/habitat preferences

#### Tidal strength preferences

#### Wave exposure preferences

#### Other preferences

### Additional Information

## Species composition

### Species found especially in this biotope

- *Haploops tubicola*

### Rare or scarce species associated with this biotope

-

### Additional information



## Sensitivity review

### Sensitivity characteristics of the habitat and relevant characteristic species

SS.SSa.IFiSa.TbAmPo is a sublittoral biotope occurring in moderately exposed and sheltered areas experiencing weak and very weak tidal streams (Connor *et al.*, 2004). These sheltered conditions support large populations of semi-permanent tube-building amphipods and polychaetes, such as *Polydora ciliata*, *Spiophanes bombyx* and *Pygospio elegans*. Additionally, the biotope is typically dominated by amphipod *Crassikorophium crassicorne* (syn. *Corophium crassicorne*), occurring with other tube-building amphipods such as *Ampelisca* sp.. These tube-building species are considered to form a key component of the biotope through stabilizing the sediment and encouraging faunal diversity, which contribute to species richness and diversity. Loss of the tube-building and dominating species would probably result in a much more uniform habitat dominated by infaunal deposit feeding polychaetes, and probably have an effect on the trophic link between these benthic prey species and the demersal fish which are strong predators. *Polydora ciliata*, *Spiophanes bombyx*, *Corophium* and *Ampelisca* spp. are therefore considered important characterizing species and are the focus of this sensitivity assessment.

### Resilience and recovery rates of habitat

*Polydora* is a small, sedentary, burrowing polychaete worm up to 3 cm long. All *Polydora* spp. make a U-shaped tube from small particles (Hayward & Ryland, 1995b). *Polydora ciliata* usually burrows into substrata containing calcium carbonate such as limestone, chalk and clay, as well as shells or oysters, mussels and periwinkles (Fish & Fish, 1996). The sexes are separate and breeding has been recorded in spring in a number of locations. In northern England, it has been recorded to occur from February until June and three or four generations succeed one another during the spawning period (Gudmundsson, 1985). Eggs are laid in a string of capsules that are attached by two threads to the wall of the burrow (Fish & Fish, 1996). After a week the larvae emerge and are believed to have a pelagic life of 2-6 weeks before settling. Length of life is no more than 1 year (Fish & Fish, 1996). Almeda *et al.* (2009) suggested low filtration rates and low growth rates despite high food availability for *Polydora ciliata* larvae, which suggested a compromise to ensure efficient larval dispersion. Larvae are substratum specific, selecting rocks according to their physical properties or sediment depending on particle size. Larvae of *Polydora ciliata* have been collected as far as 118 km offshore (Murina, 1997). Adults of *Polydora ciliata* produce a 'mud' resulting from the perforation of soft rock substrata and the larvae of the species settle preferentially on substrata covered with mud (Lagadeuc, 1991).

A *Polydora* biotope is likely to reach maturity very rapidly because *Polydora ciliata* is a short lived species that reaches maturity within a few months and has three or four spawnings during a breeding season of several months. The early reproductive period of *Polydora ciliata* often enables the species to be the first to colonize available substrata (Green, 1983). For example, in colonization experiments in Helgoland (Harms & Anger, 1983), *Polydora ciliata* settled on panels within one month in the spring. The tubes built by *Polydora* sometimes agglomerate to form layers of mud up to an average of 20 cm thick.

The settling of the first generation in April is followed by the accumulation and active fixing of mud continuously up to a peak during the month of May. The following generations do not produce a heavy settlement due to interspecific competition and heavy mortality of the larvae (Daro & Polk, 1973). Later in the year, the surface layer cannot hold the lower layers of the mud mat in place. They crumble away and are then swept away by water currents. The empty tubes of *Polydora* may

saturate the sea in June.

Other polychaetes in the biotope are likely to also recolonize disturbed areas rapidly. For example, *Spiophanes* are species with opportunistic life strategies (small size, rapid maturation and short lifespan of 1-2 years with production of large numbers of small propagules) include the polychaetes *Spiophanes bombyx* and *Spio filicornis*. Two years after dredging, abundances of opportunistic species were generally elevated relative to pre-dredging levels while communities had become numerically dominated (50-70 %) by *Spiophanes bombyx* (Gilkinson *et al.*, 2005). Van Dalfsen *et al.* (2000) found that polychaetes recolonized a dredged area within 5-10 months (reference from Boyd *et al.*, 2005), with biomass recovery predicted within 2-4 years. *Spiophanes bombyx* is regarded as a typical 'r' selecting species with a short lifespan, high dispersal potential and high reproductive rate (Niermann *et al.*, 1990). It is often found at the early successional stages of variable, unstable habitats that it is quick to colonize following perturbation (Pearson & Rosenberg, 1978). Its larval dispersal phase may allow the species to colonize remote habitats. McLusky *et al.* (1983) examined the effects of bait digging on blow lug populations in the Forth Estuary. Dug and infilled areas and unfilled basins left after digging repopulated within 1 month, whereas mounds of dug sediment took longer and showed a reduced population. Basins accumulated fine sediment and organic matter and showed increased population levels for about 2-3 months after digging. Overall recovery is generally regarded as rapid. *Pygospio elegans* were significantly depleted for >100 days after harvesting (surpassing the study monitoring timeline) and *Scoloplos armiger* demonstrated recovery >50 days after harvesting in muddy sands (Ferns *et al.*, 2000). In summary, these studies suggest recovery from fisheries pressures occurs in 4 months to >3 years depending upon the harvesting method (such as hand digging or mechanical dredging) and the size of the area impacted. As a tube building polychaete, *Pygospio elegans* aids stabilisation of sediments following disturbance. Recolonization and hence recovery may be aided by bed load transport of juvenile polychaetes and bivalves. Recolonization of *Pygospio elegans* was observed in 2 weeks by Dittmann *et al.* (1999) following a 1 month long defaunation of the sediment.

Little information was available for *Crassikorophium crassicorne* (syn. *Corophium crassicorne*) so the closely related *Corophium volutator* is here considered as an example representing *Corophium* spp. In this biotope. *Corophium volutator* is a mud shrimp with a long slender body up to 11 mm in length. The amphipod occupies semi-permanent U-shaped burrows up to 5 cm deep (Meadows & Reid, 1966) in the fine sediments of mud flats, saltmarsh pools and brackish ditches. It lives for a maximum of one year (Hughes, 1988) and females can have 2-4 broods in a lifetime (Conradi & Depledge, 1999). Populations in southerly areas such as the Dovey Estuary, Wales or Starrs Point, Nova Scotia have two reproductive episodes per year. Those populations in colder, more northerly areas such as the Ythan Estuary, Scotland or in the Baltic Sea only have one (Wilson & Parker, 1996).

On the west coast of Wales, breeding takes places from April to October and mating takes place in the burrow. Adult males crawl over the surface of the moist sediment as the tide recedes in search of burrows occupied by mature females. *Corophium volutator* forms an important food source for several species of birds and mobile predators such as fish and crabs (Hughes, 1988; Jensen & Kristensen, 1990; Raffaelli *et al.*, 1991; Flach & De Bruin, 1994; Brown *et al.*, 1999), so this behaviour makes them vulnerable to predation (Fish & Mills, 1979; Hughes, 1988; Forbes *et al.*, 1996). The females can produce 20-52 embryos in each reproductive episode (Fish & Mills 1979; Jensen & Kristensen, 1990). Juveniles are released from the brood chamber after about 14 days, and development is synchronized with spring tides, possibly to aid dispersal. Recruitment occurs within a few centimetres of the parent, although they may disperse later by swimming (Hughes, 1988). In the warmer regions where *Corophium volutator* is found, juveniles can mature in 2 months

(Fish & Mills, 1979) and add their own broods to the population. The juveniles born in May undergo rapid growth and maturation to reproduce from July to September and generate the next overwintering population (Fish & Mills, 1979).

*Corophium volutator* is one of the most abundant organisms in estuarine mudflats reaching densities of 100,000 m<sup>-2</sup> in the Stour Estuary, Suffolk (Hughes, 1988). Densities vary with geographical region and season, having been reported to rise considerably during the summer months in Gullmarsfjorden, Wadden Sea, and in the Crouch Estuary in southeast England (Flach & De Bruin, 1993; Gerdol & Hughes, 1993).

The amphipod genus *Ampelisca* has some life history traits that allow them to recover quickly where populations are disturbed. They do not produce large numbers of offspring but reproduce regularly and the larvae are brooded, giving them a higher chance of survival within a suitable habitat than free-living larvae. *Ampelisca* has a short lifespan and reaches sexual maturity in a matter of months allowing a population to recover abundance and biomass in a very short period of time (MES, 2008). Experimental studies have shown *Ampelisca abdita* to be an early colonizer, in large abundances of defaunated sediments where local populations exist to support recovery (McCall, 1977) and *Ampelisca abdita* have been shown to migrate to, or from, areas to avoid unfavourable conditions (Nichols & Thompson, 1985). *Ampelisca* spp. are very intolerant of oil contamination and the recovery of then *Ampelisca* populations in the fine sand community in the Bay of Morlaix took up to 15 years following the *Amoco Cadiz* oil spill, probably due to the amphipods' low fecundity, lack of pelagic larvae and the absence of local unperturbed source populations (Poggiale & Dauvin, 2001). Mills (1967) reported that *Ampelisca vadorum* and *Ampelisca abdita* produced only 1 brood per generation but there were 2 or more generations per year. In the English Channel, two reproductive patterns were identified. Species such as *Ampelisca tenuicornis* and *Ampelisca typica* produced two generations per year. The juveniles born in May-June were able to brood in September-October (Dauvin, 1988b; Dauvin, 1988c). Species such as *Ampelisca armoricana* and *Ampelisca sarsi* produced only one brood per generation and per year (Dauvin, 1989; Dauvin, 1988d). *Ampelisca brevicornis* showed an intermediate cycle with one generation per year during cold years (cold spring) and two generations per year during warm years (warm spring) and its cycle is intermediate between univoltine cycle and bivoltine cycle (Dauvin, 1988b,c,d,e; Dauvin, 1989; Dauvin & Bellan-Santini, 1990).

**Resilience assessment.** Removal of the characterizing species would result in the biotope being lost and/or reclassified. The tube building polychaetes such as *Polydora ciliata* and *Pygospio elegans* generally disperse via pelagic larvae (Fish & Fish, 1996) and therefore recruitment may occur from distant populations. These are likely to recolonize disturbed areas first, although the actual pattern will depend on recovery of the habitat, season of occurrence and other factors. Dispersal of the infaunal deposit feeders, such as *Scoloplos armiger* and *Arenicola marina*, occurs through burrowing of the benthic larvae and adults (Beukema & De Vlas, 1979; Fish & Fish, 1996). Recruitment must therefore occur from local populations or by longer distance dispersal during periods of bedload transport. Recruitment is therefore likely to be predictable if local populations exist but patchy and sporadic otherwise. The life cycles of amphipods varies between the different families. Based on the intertidal species, *Corophium volutator*, the *Corophium* sp. may produce several broods over the summer breeding season (Fish & Fish, 1996), but there is no larval stage. Dispersal is limited to local movements of these sub-juveniles and migration of the adults and hence recruitment is limited by the presence of local, unperturbed source populations (Poggiale & Dauvin, 2001). Dispersal of sub-juveniles may be enhanced by the brooding females leaving their tubes and swimming to uncolonized areas of substratum before the eggs hatch (Mills, 1967). So, where perturbation removes a portion of the population or even causes local extinction (resistance High,

Medium or Low) resilience is likely to be **High** for as long as recruitment from neighbouring areas and/or adult migration is possible. However, even in areas of suitable habitat that are isolated, where total extinction of the population occurs (resistance None) recovery is likely to depend on favourable hydrodynamic conditions that will allow recruitment from farther away. Given the low energy environment where the biotope occurs, recruitment to recolonize impacted area may take longer. However, once an area has been recolonized, restoration of the biomass of the characterizing species is likely to occur quickly and resilience is likely to be **Medium** (full recovery within 2-10 years).

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

## Hydrological Pressures

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

Murina (1997) categorized *Polydora ciliata* as a eurythermal species because of its ability to spawn in temperatures ranging from 10.6-19.9°C. This is consistent with a wide distribution in north-west Europe, which extends into the warmer waters of Portugal and Italy (Pardal *et al.*, 1993; Sordino *et al.*, 1989). In the western Baltic Sea, Gulliksen (1977) recorded high abundances of *Polydora ciliata* in temperatures of 7.5 to 11.5°C and in Whitstable in Kent sea temperatures varied between 0.5 and 17°C (Dorsett, 1961). Growth rates may increase if temperature rises. For example, at Whitstable in Kent, Dorsett (1961) found that a rapid increase in growth coincided with the rising temperature of the seawater during March.

Amphipods, however, are reported to have low tolerance to temperature changes (Bousfield, 1973) although lethal limits are not given. However, the amphipods that occur within this habitat are mobile and can avoid unfavourable conditions to some extent. Furthermore, *Corophium volutator* is widely distributed in the north Atlantic, American and European coasts; from western Norway to the Mediterranean and the Black Sea and Azov Sea (Neal & Avant, 2006). The amphipod is subject to temperatures of 1°C in the winter to 17°C in the summer (Wilson & Parker, 1996) but can resist much higher temperatures (Meadows & Ruagh, 1981). Temperature increases may lead to indirect effects on populations of the characterizing species, through changes in the distribution and prevalence of parasites and pathogens (see microbial pathogens pressure). Using a simulation model, Mouritsen *et al.* (2005) demonstrated that a 3.8 °C increase in ambient temperature would probably result in a parasite-induced collapse of the amphipod population in the Wadden Sea. This indirect effect is assessed in the introduction of microbial pathogens pressure.

Other polychaetes species in the biotope, e.g. *Scoloplos armiger* and *Pygospio elegans*, show a relationship between timing of reproduction and temperature. Studies on the polychaete *Scoloplos armiger* in the Wadden Sea (North Sea) displayed that intertidal 'Type I' *Scoloplos armiger* reproduced in spring, through holobenthic development, triggered by a rise in seawater temperature above 5°C (Kruse *et al.*, 2004). Gibson & Harvey (2000), in a study on asexual reproduction of *Pygospio elegans* in Nova Scotia, Canada, found temperature did not influence reproduction strategy (planktotrophy, lecithotrophy or asexual reproduction) but that environmental conditions, including temperature, influenced timing of reproduction. Furthermore, *Pygospio elegans* has been recorded in seas with a temperature range of 1.6°C to 12.5°C (OBIS, 2016). *Scoloplos armiger* occurs in seas with a temperature range of between 8.8°C and 13°C (OBIS, 2016). Both *Scoloplos armiger* and *Pygospio elegans* tolerate a wide temperature range although optimal temperature ranges, based on feeding and reproductive success are more restricted, suggesting a 5°C increase in temperature over a month may increase temperatures above the preferred range, but not cause widespread mortality. These temperature ranges are not derived from peer reviewed studies and therefore caution should be used with the interpretation.

No information was found regarding the intolerance of *Spiophanes bombyx* to temperature. *Spiophanes bombyx* is found in the Mediterranean (Hayward & Ryland, 1995b), which is likely to be warmer than the waters around Britain and Ireland.

**Sensitivity assessment.** Typical surface water temperatures around the UK coast vary seasonally from 4-19°C (Huthnance, 2010). It is possible that reproduction may be affected, but no mortality is expected and the characterizing species of this biotope are likely to be able to resist a long-term increase in temperature of 2°C and may resist a short-term increase of 5°C. Resistance and resilience are therefore assessed as **High** and the biotope assessed as **Not Sensitive** at the benchmark level.

#### Temperature decrease (local)

**High**

Q: Medium A: Medium C: Medium

**High**

Q: High A: High C: High

**Not sensitive**

Q: Medium A: Medium C: Medium

Murina (1997) categorized *Polydora ciliata* as a eurythermal species because of its ability to spawn in temperatures ranging from 10.6-19.9°C. This is consistent with a wide distribution in north-west Europe. In the western Baltic Sea, Gulliksen (1977) recorded high abundances of *Polydora ciliata* in temperatures of 7.5 to 11.5°C and in Whitstable in Kent abundance was high when winter water temperatures dropped to 0.5°C (Dorsett, 1961). During the extremely cold winter of 1962/63 *Polydora ciliata* was apparently unaffected (Crisp, 1964).

Amphipods, however, are reported to have low tolerance to temperature changes (Bousfield, 1973) although lethal limits were not given. Mills (1967) reported that gonadal growth of *Ampelisca vadorum* and *Ampelisca abdita* is retarded by low temperatures, thus delaying maturity, and feeding rate was reduced below 10°C. *Corophium volutator* is subject to temperatures of 1°C in the winter to 17°C in the summer (Wilson & Parker, 1996), but the population may equally reduce activity and delay reproduction if the temperature drops below 7°C. Sudden pulses of very cold water can disrupt the circa-tidal rhythms of *Corophium volutator* by resetting the onset of swimming behaviour. For example, a 6 hour cold spell would lead to the population trying to swim at low tide and leave them vulnerable to increased predation. However, it took temperatures of 15-20°C below ambient temperature to induce this response (Holmström & Morgan, 1983b). Furthermore, Drolet *et al.* (2013) sampled two intertidal mudflats in the upper Bay of Fundy, Canada, over two consecutive winters (2009–2011), where sediment temperature, 5 cm deep,

reached  $2^{\circ}\text{C}$ . The authors' concluded *Corophium volutator* survived winter through physiological adaptations. Additionally, Crisp (1964) reported that species of amphipod and isopods seemed to be unharmed by the severe winter of 1962-1963, which may be due to burial in sediments buffering temperature or seasonal migration to deeper waters to avoid freezing.

Other polychaete species that are typically common in the biotope, *Scoloplos armiger* and *Pygospio elegans*, show a relationship between timing of reproduction and temperature. Studies on the polychaete *Scoloplos armiger* in the Wadden Sea (North Sea) displayed that intertidal 'Type I' *Scoloplos armiger* reproduce in spring, through holobenthic development triggered by a rise in seawater temperature above  $5^{\circ}\text{C}$  (Kruse *et al.*, 2004). Timing of reproduction of *Pygospio elegans* in a study from the Baltic Sea was linked to environmental conditions including temperature (Anger, 1984). No information was found regarding the intolerance of *Spiophanes bombyx* to temperature. However, *Spiophanes bombyx* is found in waters off Denmark (Thorson, 1946) which are likely to be colder than British and Irish waters.

**Sensitivity assessment.** Typical surface water temperatures around the UK coast vary seasonally from  $4\text{-}19^{\circ}\text{C}$  (Huthnance, 2010). It is possible that reproduction may be affected, but no mortality is expected and the characterizing species of this biotope are likely to be able to resist a long-term decrease in temperature of  $2^{\circ}\text{C}$  and may resist a short-term decrease of  $5^{\circ}\text{C}$ . Resistance and resilience are therefore assessed as **High** and the biotope judged as **Not Sensitive** at the benchmark level.

#### Salinity increase (local)

Low

Q: Low A: NR C: NR

High

Q: High A: High C: High

Low

Q: Low A: NR C: NR

*Polydora ciliata* is a euryhaline species inhabiting fully marine and estuarine habitats. However, there are no records of the species or the biotope occurring in hypersaline waters.

Monitoring at a Spanish desalination facility where discharges close to the outfall reached a salinity of 53, found that amphipods, including *Ampelisca* spp. were sensitive to the increased salinity and that species free-living in the sediment were most sensitive (De-la-Ossa-Carretero *et al.*, 2016). *Corophium volutator* occupies a greater variety of habitats and is found in fine sediments of mud flats, saltmarsh pools and brackish ditches. It is therefore exposed in its natural range to salinity extremes, from freshwater flushing during periods of high rainfall and increases in salinity during periods of high evaporation. *Corophium volutator* is, therefore, an exceptionally euryhaline species able to tolerate 2-50 psu (McLusky, 1968) but growth is fastest at 15-20 psu (McLusky, 1967; McLusky, 1970 cited in Meadows & Ruagh, 1981). The interstitial salinity is more important for *Corophium volutator* than that of the overlying water. Sustained periods of increased salinity are required to alter that of the interstitial water and there is a lag between salinity changes and the response of *Corophium volutator* (McLusky, 1968). Sudden increases in salinity delay swimming activity (Harris & Morgan, 1984a). *Corophium volutator* will also migrate from areas of unfavourable salinity (McLusky, 1968).

Other polychaete species in the biotope are likely to tolerate increases in salinity. In the western Baltic Sea, *Scoloplos armiger* abundance was greatest between 12 psu and 17 psu and reduced abundance with increasing salinity was observed (Gogina *et al.*, 2010). As *Scoloplos armiger* is a species complex and is not a cosmopolitan species there may be inconsistencies between general environmental settings found in literature and observed and predicted distribution limits within study sites (Bleidorn *et al.*, 2006 cited in Gogina *et al.*, 2010). *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 at the pressure benchmark level. However, an increase in one MNCR salinity category above the usual range of the biotope may reduce abundance as both species are most abundant in variable and fully marine salinity categories (and *Scoloplos armiger* has displayed negative responses to increasing salinity). No information was found concerning the reaction of *Spiophanes bombyx* to hypersaline conditions (>40 psu). It is unlikely that *Spiophanes bombyx* would experience hypersaline conditions, therefore unlikely to be adapted to such conditions.

**Sensitivity assessment.** The characterizing species of this biotope are euryhaline and likely to be resistant of an increase in salinity. However, the biotope occurs in full saline conditions (Connor *et al.*, 2004) and is unlikely to experience hypersaline conditions. A chronic increase at the pressure benchmark level that remains over time is likely to result in the death of a portion of the individuals in the population, particularly of *Polydora ciliata* due to its inability to move to more favourable conditions. Resistance is therefore assessed as **Low**, but with low confidence. Resilience is likely to be **High**, so the biotope is considered to have **Low** sensitivity to an increase in salinity at the pressure benchmark level.

#### Salinity decrease (local)

**High**

Q: High A: High C: High

**High**

Q: High A: High C: High

**Not sensitive**

Q: High A: High C: High

*Polydora ciliata* is a euryhaline species inhabiting fully marine and estuarine habitats. In an area of the western Baltic Sea, where bottom salinity was between 11.1 and 15.0 psu *Polydora ciliata* was the second most abundant species with over 1000 individuals per m<sup>2</sup> (Gulliksen, 1977).

*Corophium volutator* is an exceptionally euryhaline species able to tolerate 2-50 psu (McLusky, 1968) but growth is fastest at 15-20 psu (McLusky, 1970 cited in Meadows & Ruagh, 1981). *Corophium volutator* is a hyperosmotic regulator and the tolerance of its tissues is 13-50 psu but it needs a salinity of above 5 psu in order to moult, since osmoregulation is lost during moulting (McLusky, 1967). A salinity of at least 7.5 psu is required for reproduction (McLusky, 1968). Females undergoing a pre-copulatory moult failed to lay eggs below salinities of 3‰, with the lowest salinity at which all females moulted and laid eggs was 20‰ (Mills & Fish, 1980). Changes in salinity are very unlikely to cause mortality but may alter population distribution and abundances within the biotope as the species is likely to move to more favourable conditions or to suffer some effects on reproduction with resulting decreases in abundance.

Other polychaete species in the biotope are likely to tolerate decreases in salinity. *Scoloplos armiger* shows a lower salinity limit of 10.5 psu (Gogina *et al.*, 2010), suggesting the species is tolerant of a decrease in salinity. *Pygospio elegans* was 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). *Spiophanes bombyx* is a euryhaline species (Bailey-Brook, 1976; Maurer & Lethem, 1980), inhabiting fully saline and estuarine habitats.

**Sensitivity assessment.** SS.SSa.IFiSa.TbAmPo occurs in full salinity conditions (Connor *et al.*, 2004). However, the evidence presented indicate that the species indicative of sensitivity occur in environments of low salinities, and are therefore likely to resist a decrease in salinity at the

pressure benchmark level (to variable 18-35 ppt). Resistance is assessed as **High** and resilience as **High** (by default), so the biotope is considered **Not Sensitive** to a decrease in salinity at the pressure benchmark level.

### Water flow (tidal current) changes (local)

**Low**

Q: Medium A: Low C: High

**High**

Q: High A: High C: High

**Low**

Q: Medium A: Low C: High

Changes in water flow are likely to change the sediment characteristics in which the biotope occurs, primarily by resuspending and preventing deposition of finer particles (Hiscock, 1983). A decrease in water flow is unlikely to cause any impact on the biotope as species are adapted to incremental deposition, typical of low energy environments such as those where the biotope occurs. On the other hand however, an increase would likely result in a decrease in tube-building material for the characterizing species, and the lack of deposition of particulate matter at the sediment surface would reduce food availability for the deposit feeders in the biotope. The resultant energetic cost over one year would be likely to result in some mortality of tube-builders and infauna. For example, *Polydora ciliata* was present and colonized test panels in Helgoland in three areas; two exposed to strong tidal currents and one site sheltered from currents (Harms & Anger, 1983). However, very strong water flows may sweep away *Polydora* colonies, often in a thick layer of mud on a hard substratum. Small *Corophium volutator* cannot resettle after swimming at current speeds approx. 0.1 m/s (Ford & Paterson, 2001), which probably explains why they mainly swim at high tide (Hughes, 1988). An increase in water flow rate could cause swimming *Corophium volutator* to be swept away from suitable habitat and cause high mortality. The species has been reported as not using flow velocity as a cue to stimulate swimming (Ford & Paterson, 2001).

The most damaging effect of increased flow rate would be the erosion of the medium to fine muddy sand substratum as this could eventually lead to loss of the habitat. Emergent species, such as the *Polydora ciliata* tubes that characterize this biotope, may create turbulent flow leading to particle resuspension. However, the medium to fine muddy sands offer increased cohesiveness and resistance to erosion and, because of their high numbers, amphipods are thought to stabilize the intertidal sediments in which they reside (Mouritsen *et al.*, 1998).

Additionally, where a change in water flow rate changes sediment characteristics, with increased deposits of coarser sediments, characterizing species may no longer be supported due to particular substratum preferences. For example, *Spiophanes bombyx* preferred substratum is fine sands, and *Scoloplos armiger* and *Pygospio elegans* thrive in medium particle size, therefore a change in sediment characteristics may result in a reduced distribution and extent of the populations.

**Sensitivity assessment.** Sand particles are most easily eroded and likely to be eroded at about 0.20 m/s (based on Hjulström-Sundborg diagram, Sundborg, 1956). This biotope occurs on medium to very fine muddy sand (Connor *et al.*, 2004), and although having a smaller grain size than sand, the very fine muddy sands will require greater critical erosion velocities because of their cohesiveness. SS.SSa.IFiSa.TbAmPo is recorded in weak (>0.5 m/s) and very weak (negligible) tidal streams (Connor *et al.*, 2004). Although a decrease in water flow rate is likely to be irrelevant, an increase in water flow at the pressure benchmark is likely to result in loss of the characterizing *Corophium* spp., based on reports of *Corophium volutator* being unable to resettle after swimming at current speeds approx. 0.1 m/s (Ford & Paterson, 2001). Furthermore, stronger currents are likely to wash away the community of semi-permanent tube-building amphipods and polychaetes that characterize the biotope. Resistance is therefore assessed as **Low** and resilience as **High** and the



biotope judged as having **Low** sensitivity to a change in water flow at the pressure benchmark level.

### Emergence regime changes

**High**

Q: High A: High C: High

**High**

Q: High A: High C: High

**Not sensitive**

Q: High A: High C: High

SS.SSa.IFiSa.TbAmPo occurs in the shallow subtidal fringe and a change in emergence could potentially affect the upper extent of shallow examples of the biotope that are likely to be emerged on extreme low tides. All characterizing species would probably survive an increase in emergence. However, the species can only feed when immersed and therefore likely to experience reduced feeding opportunities. Over the course of a year, the resultant energetic cost is likely to cause some mortality. In addition, increased emergence is likely to increase the vulnerability to predation from shore birds, particularly of *Corophium* spp. which form an important food source for several species of birds and mobile predators (Hughes, 1988; Jensen & Kristensen, 1990; Raffaelli *et al.*, 1991; Flach & De Bruin, 1994; Brown *et al.*, 1999). A decrease in emergence is likely to allow the biotopes to extend their upper limits, where suitable substrata exist.

**Sensitivity assessment.** Species within the biotope exhibit physiological and behavioural adaptations to changes in emergence and the biotope as a group is unlikely to be very sensitive. Furthermore, although some mortality of the characterizing species is likely to occur, it is considered that populations in the mid-range examples of the biotope are likely to be unaffected by changes in the emergence regime. Resistance and resilience (by default) are therefore assessed as **High**, and the biotope considered **Not Sensitive**.

### Wave exposure changes (local)

**High**

Q: Low A: NR C: NR

**High**

Q: High A: High C: High

**Not sensitive**

Q: Low A: NR C: NR

Where the biotope occurs in the shallow subtidal, it is likely to be affected by winter storms. Storms may cause dramatic changes in distribution of macro-infauna by washing out dominant species, opening the sediment to recolonization by adults and/or available spat/larvae (Eagle, 1975; Rees *et al.*, 1976; Hall, 1994) and by reducing success of recruitment by newly settled spat or larvae (see Hall, 1994 for review). However, the biotope is found in moderately exposed and sheltered sites (Connor *et al.*, 2004), which is likely to be an important factor in allowing for the development of the characteristic sedimentary regime and associated fauna that characterize this biotope.

Feeding of the characterizing species may be impaired in strong wave action and changes in wave exposure may also influence the supply of particulate matter for tube-building polychaetes and amphipods. For example, Mills (1967) reported that *Ampelisca* flats in Barnstable, USA, were damaged noticeably by winter storms. Decreases in wave exposure may influence the supply of particulate matter because wave action may have an important role in resuspending the sediment that is required by the species to build their tubes. Food supplies may also be reduced affecting growth and fecundity of the species.

Potentially the most damaging effect of increased wave heights would be the erosion of the medium to fine muddy sand substrata as this could eventually lead to loss of the habitat. Increased wave action may disturb the mud in which *Corophium volutator* lives and make it impossible for them to maintain burrows and may affect their ability to settle after swimming. Increased erosion would lead to the loss of habitat and removal of the characterizing species.

**Sensitivity assessment.** Some erosion will occur naturally and storm events may be more significant in loss and damage of fine muddy sands that characterize the biotope. The biotope occurs in moderately exposed and sheltered conditions (Connor *et al.*, 2004), and a change at the benchmark level is likely to fall within the range experienced by the mid-range examples of this biotope. The biotope is therefore considered to have **High** resistance to changes at the pressure benchmark, where these do not lead to increased erosion of the substratum. Resilience is therefore assessed as **High** and the biotope is considered to be **Not Sensitive**, at the pressure benchmark.

## Chemical Pressures

	Resistance	Resilience	Sensitivity
<b>Transition elements &amp; organo-metal contamination</b>	Not Assessed (NA) Q: NR A: NR C: NR	Not assessed (NA) Q: NR A: NR C: NR	Not assessed (NA) Q: NR A: NR C: NR

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

Experimental studies with various species suggest that polychaete worms are quite resistant to heavy metals (Bryan, 1984). *Polydora ciliata* occurred in an area of the southern North Sea polluted by heavy metals but was absent from sediments with very high heavy metal levels (Diaz-Castaneda *et al.*, 1989).

For most metals, toxicity to crustaceans increases with decreased salinity and elevated temperature, therefore marine species living within their normal salinity range may be less susceptible to heavy metal pollution than those living in salinities near the lower limit of their salinity tolerance (McLusky *et al.*, 1986). In laboratory investigations, Hong & Reish (1987) observed 96 hour LC<sub>50</sub> water column concentrations of between 0.19 and 1.83 mg/l for several species of amphipod. *Corophium volutator* is highly intolerant of metal pollution at levels often found in estuaries from industrial outfalls and contaminated sewage. A concentration 38 mg Cu/l was needed to kill 50% of *Corophium volutator* in 96 hour exposures (Bat *et al.*, 1998). Other metals are far more toxic to *Corophium volutator*, e.g. zinc is toxic over 1 mg/l and toxicity to metals increases with increasing temperature and salinity (Bryant *et al.*, 1985). Mortality of 50% is caused by 14 mg/l (Bat *et al.*, 1998). Although exposure to zinc may not be lethal, it may affect the perpetuation of a population by reducing growth and reproductive fitness. Mercury was found to be very toxic to *Corophium volutator*, e.g. concentrations as low as 0.1 mg/l caused 50% mortality in 12 days. Other metals known to be toxic include cadmium, which causes 50% mortality at 12 mg/l (Bat *et al.*, 1998); and arsenic, nickel and chromium which are all toxic over 2 mg/l (Bryant *et al.*, 1984; Bryant *et al.*, 1985a; 1985).

<b>Hydrocarbon &amp; PAH contamination</b>	Not Assessed (NA) Q: NR A: NR C: NR	Not assessed (NA) Q: NR A: NR C: NR	Not assessed (NA) Q: NR A: NR C: NR
--	--	--	--

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

Generally soft sediment inhabitants, especially infaunal polychaetes, are particularly affected by oil pollution (Suchanek, 1993). For example, Jacobs (1980) investigated the effects of the *Amoco Cadiz* oil spill in 1978 and noted that the numbers of spionidae polychaetes decreased after the

spill. In analysis of kelp holdfast fauna following the *Sea Empress* oil spill in Milford Haven the fauna present, including *Polydora ciliata*, showed a strong negative correlation between numbers of species and distance from the spill (SEEEC, 1998). After the extensive oil spill in West Falmouth, Massachusetts, Grassle & Grassle (1974) followed the settlement of polychaetes in the disturbed area. Species with the most opportunistic life histories, including *Polydora ligni*, were able to settle in the area. This species has some brood protection which enables larvae to settle almost immediately in the nearby area (Reish, 1979). Furthermore, Gray *et al.* (1990) found that *Scoloplos armiger* was a dominant species in uncontaminated soft sediments at a case study site adjacent to the Ekofisk oil field but were not present at contaminated sites, suggesting *Scoloplos armiger* are also intolerant to hydrocarbon contaminants.

Amphipods in general, and ampeliscid amphipods in particular seem particularly intolerant of contamination with oil. Dauvin (1998) reported reductions in abundance, biomass and production of *Ampelisca* sp. following the *Amoco Cadiz* oil spill. Furthermore, light fractions (C10 - C19) of oils are much more toxic to *Corophium volutator* than heavier fractions (C19 - C40). In exposures of up to 14 days, light fraction concentrations of 0.1 g/kg sediment caused high mortality. It took 9 g/kg sediment to achieve similar mortalities with the heavy fraction (Brils *et al.*, 2002). In the Forth Estuary, *Corophium volutator* was excluded for several hundred metres around the outfalls from hydrocarbon processing plants. Roddie *et al.* (1994) found high levels of mortality of *Corophium* at sites contaminated with crude oil.

#### Synthetic compound contamination

Not Assessed (NA)

Q: NR A: NR C: NR

Not assessed (NA)

Q: NR A: NR C: NR

Not assessed (NA)

Q: NR A: NR C: NR

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

In general, crustaceans are widely reported to be intolerant of synthetic chemicals (Cole *et al.*, 1999) and intolerance to some specific chemicals has been observed in amphipods. Species of a different genus are likely to differ in their susceptibility to synthetic chemicals and that this may be related to differences in their physiology (Powell, 1979). *Corophium volutator* is paralysed by pyrethrum based insecticide sprayed onto the surface of the mud (Gerdol & Hughes, 1993) and pyrethrum would probably cause significant mortalities if it found its way into estuaries from agricultural runoff. Nonylphenol is an anthropogenic pollutant that regularly occurs in water bodies, it is an oestrogen mimic that is produced during the sewage treatment of non-ionic surfactants and can affect *Corophium volutator* (Brown *et al.*, 1999). Nonylphenol is a hydrophobic molecule and often becomes attached to sediment in water bodies. This will make nonylphenol available for ingestion by *Corophium volutator* in estuaries where much of the riverine water-borne sediment flocculates and precipitates out of suspension to form mudflats. Nonylphenol is not lethal to *Corophium volutator* but does reduce growth and has the effect of causing the secondary antennae of males to become enlarged which can make the amphipods more vulnerable to predators (Brown *et al.*, 1999). *Corophium volutator* is killed by 1% ethanol if exposed for 24 hours or more but can withstand higher concentrations in short pulses. Such short pulses, however, have the effect of rephasing the diel rhythm and will delay the timing of swimming activity for the duration of the ethanol pulse (Harris & Morgan, 1984b).

The anti-parasite compound ivermectin is highly toxic to benthic polychaetes and crustaceans (Black *et al.*, 1997; Collier & Pinn, 1998; Grant & Briggs, 1998, cited in Wilding & Hughes, 2010). OSPAR (2000) stated that, at that time, ivermectin was not licensed for use in mariculture but was incorporated into the feed as a treatment against sea lice at some farms. Ivermectin has the

potential to persist in sediments, particularly fine-grained sediments at sheltered sites. Data from a farm in Galway, Ireland indicated that ivermectin was detectable in sediments adjacent to the farm at concentrations up to 6.8 µm/kg and to a depth of 9 cm (reported in OSPAR, 2000). Infaunal polychaetes have been affected by deposition rates of 78-780 mg ivermectin/m<sup>2</sup>. Furthermore, *Polydora ciliata* was abundant at polluted sites close to acidified, halogenated effluent discharge from a bromide-extraction plant in Amlwch, Anglesey (Hoare & Hiscock, 1974). Spionid polychaetes were found by McLusky (1982) to be relatively resistant of distilling and petrochemical industrial waste in Scotland.

#### Radionuclide contamination

No evidence (NEv)

Q: NR A: NR C: NR

No evidence (NEv)

Q: NR A: NR C: NR

No evidence (NEv)

Q: NR A: NR C: NR

*Corophium volutator* readily absorbs radionuclides such as americium and plutonium from water and contaminated sediments (Miramand *et al.*, 1982). However, the effect of contamination of the individuals was not known but accumulation through the food chain was assumed (Miramand *et al.*, 1982).

There was **No Evidence** on which to base an assessment.

#### Introduction of other substances

Not Assessed (NA)

Q: NR A: NR C: NR

Not assessed (NA)

Q: NR A: NR C: NR

Not assessed (NA)

Q: NR A: NR C: NR

This pressure is **Not assessed**.

#### De-oxygenation

Low

Q: High A: Medium C: High

High

Q: High A: High C: High

Low

Q: High A: Medium C: High

*Polydora ciliata* is repeatedly found at localities with oxygen deficiency (Pearson & Rosenberg, 1978). For example, in polluted waters in Los Angeles and Long Beach harbours *Polydora ciliata* was present in the oxygen range 0.0-3.9 mg/l and the species was abundant in hypoxic fjord habitats (Rosenberg, 1977). Furthermore, in a study investigating a polychaete community in the north west Black Sea, *Polydora ciliata* was observed in all four study sites, including those severely affected by eutrophication and hypoxia as a result of the discharge of waste waters (Vorobyova *et al.*, 2008). However, *Polydora ciliata* is unlikely to be able to resist anoxic conditions. Hansen *et al.* (2002) reported near total extinction of all metazoan in the Mariager Fjord (Denmark), including *Polydora* spp. after a severe hypoxia event that resulted in complete anoxia in the water column for two weeks. Additionally, Como & Magni (2009) investigated seasonal variations in benthic communities known to be affected by episodic events of hypoxia. The authors observed that abundance of *Polydora ciliata* varied seasonally, decreasing during the summer months, and suggested it could be explained by the occurrence of hypoxic/anoxic conditions and sulphidic sediments during the summer. No details of the levels of dissolved oxygen leading to these community responses were provided. Other polychaetes in the biotope are also likely to be able to deal with hypoxia. For example, during low tide the polychaete *Scoloplos armiger* survives deoxygenation by ascending into the oxidative layer where it is able to maintain aerobic metabolism. In laboratory conditions *Scoloplos armiger* survived low oxygen conditions for 40 hours (Schöttler & Grieshaber, 1988).

Amphipods appear not to be tolerant of reduced oxygenation. For example, *Ampelisca agassizi*, is

reported to be intolerant of hypoxia (see review by Diaz & Rosenberg, 1995) and *Jassa falcata*, another tube building amphipod species, was absent from Californian harbours with low oxygen concentrations (0-2.5 mg/l). *Corophium volutator* is highly sensitive to hypoxia and suffers 50% mortality after just 4 hours in hypoxic conditions, or in 2 hours if there is rapid build-up of sulphide (Gamenick *et al.*, 1996). These conditions often occur in estuaries where drifting macroalgae (such as *Fucus* sp.) settle on the mudflats in small patches. These results are largely in concordance with other work by Gamble (1970) who found that survival rates were temperature dependent with individuals surviving longer at lower temperatures. The level of oxygen was not assessed by Gamenick *et al.* (1996) and the description of the experimental set-up suggests that anoxic test conditions were used rather than hypoxic. Gamble (1970) found that at 5°C most individuals were inactive after 30 minutes exposure to anaerobic seawater and that mortality occurred later, the inactivity may have allowed the species to survive longer (Gamble, 1970).

Riedel *et al.* (2012) assessed the response of benthic macrofauna to hypoxia advancing to anoxia in the Mediterranean. The hypoxic and anoxic conditions were created for 3-4 days in a box that enclosed in-situ sediments. Polychaetes appeared to be sensitive to hypoxia, as only 10% of polychaetes survived. In general, epifauna were more sensitive than infauna, mobile species more sensitive than sedentary species and predatory species more sensitive than suspension and deposit feeders. The test conditions did not lead to the production of hydrogen sulphide which may have reduced mortalities compared to some observations. Nierman *et al.* (1990) reported changes in a fine sand community for the German Bight in an area with regular seasonal hypoxia. In 1983, oxygen levels were exceptionally low (<3 mg O<sub>2</sub>/l) in large areas and <1 mg O<sub>2</sub>/l in some areas. Species richness decreased by 30-50% and overall biomass fell. *Spiophanes bombyx* was found in small numbers at some, but not all areas, during the period of hypoxia. Once oxygen levels returned to normal *Spiophanes bombyx* increased in abundance; the evidence suggests that at least some individuals would survive hypoxic conditions.

**Sensitivity assessment.** Cole *et al.* (1999) suggested possible adverse effects on marine species exposed to dissolved oxygen concentrations below 4 mg/l and probable adverse effects below 2 mg/l. *Polydora ciliata* is repeatedly found at localities with oxygen deficiency (Pearson & Rosenberg, 1978) and seems to only be affected by severe deoxygenation episodes. Furthermore, opportunistic *Polydora* spp. were also reported to be amongst the first to recover from hypoxic events (Hansen *et al.*, 2002; Van Colen *et al.*, 2010). Other polychaetes in the biotope are likely to behave similarly. However, mortality of *Corophium* and other amphipod species is likely to occur. Resistance to deoxygenation at the pressure benchmark level is therefore likely to be **Low** and resilience is likely to be **High**. The biotope is therefore considered to have **Low** sensitivity to exposure to dissolved oxygen concentration of less than or equal to 2 mg/l for 1 week.

#### Nutrient enrichment

Not relevant (NR)

Q: NR A: NR C: NR

Not relevant (NR)

Q: NR A: NR C: NR

Not sensitive

Q: NR A: NR C: NR

Connor *et al.* (2004) suggested that many of the sites featuring this biotope are situated near fish farms, so it is possible that it may have developed as the result of moderate nutrient enrichment. *Polydora ciliata* is often found in environments subject to high levels of nutrients. For example, the species was abundant in areas of the Firth of Forth exposed to high levels of sewage pollution (Smyth, 1968), in nutrient rich sediments in the Mondego Estuary, Portugal (Pardal *et al.*, 1993), and the coastal lagoon Lago Fusaro, Naples (Sordino *et al.*, 1989). The extensive growths of *Polydora ciliata* in mat formations were recorded at West Ganton, in the Firth of Forth, prior to the introduction of the Sewage Scheme (Read *et al.*, 1983). The abundance of the species was probably

associated with their ability to use the increased availability of nutrients as a food source and silt for tube building.

Similarly, amphipods appear to be tolerant of, and indeed prefer, high nutrient levels. However, in the Ythan Estuary, Scotland, nutrient enrichment causes the mudflats to become covered with algal mats consisting mainly of the gutweed *Ulva intestinalis*. These mats physically perturb *Corophium volutator* by preventing burrowing and normal feeding. In areas where the mats did not occur, the density of *Corophium volutator* was 11 times higher than under the algae. When the algae died-back in the winter, the areas were rapidly recolonized by *Corophium volutator* from adjacent patches where the gutweed could not grow and population growth was high from feeding on the rotting algae. In the spring, the gutweed returned and the *Corophium volutator* were excluded once again (Raffaelli *et al.*, 1991).

**Sensitivity assessment.** This pressure relates to increased levels of nitrogen, phosphorus and silicon in the marine environment compared to background concentrations. The characterizing species of this biotope are likely to be able to resist and be favoured by nutrient enrichment where increased availability of nutrients may be used as a source of food (Hiscock *et al.*, 2005a). However, where nutrient enrichment causes proliferation of algal mats, *Corophium volutator* densities are likely to be reduced as a result of smothering. Nevertheless, the biotope is considered **Not Sensitive** at the pressure benchmark level, which is set at compliance with Water Framework Directive (WFD) criteria for good status, based on nitrogen concentration (UKTAG, 2014).

## Organic enrichment

High

Q: High A: Medium C: High

High

Q: High A: High C: High

Not sensitive

Q: High A: Medium C: High

*Polydora ciliata* is often found in environments subject to high levels of nutrients. For example, the species was abundant in areas of the Firth of Forth exposed to high levels of sewage pollution (Smyth, 1968), in nutrient rich sediments in the Mondego Estuary, Portugal (Pardal *et al.*, 1993), and the coastal lagoon Lago Fusaro in Naples (Sordino *et al.*, 1989). The extensive growths of *Polydora ciliata* in mat formations were recorded at West Ganton, in the Firth of Forth, prior to the introduction of the Sewage Scheme (Read *et al.*, 1983). The abundance of the species was probably associated with their ability to use the increased availability of nutrients as a food source and silt for tube building. However, *Polydora ciliata* can also occur in organically poor areas (Pearson & Rosenberg, 1978).

In colonization experiments in an organically polluted fjord receiving effluent discharge from Oslo, *Polydora ciliata* had settled in large numbers within the first month (Green, 1983). However, Callier *et al.* (2007) investigated the spatial distribution of macrobenthos under a suspended mussel culture, in eastern Canada, where the sedimentation of organic matter to the bottom was approx. 1-3 gC/m<sup>2</sup>/day. *Polydora ciliata* was recorded as absent in the sites under the suspended mussel farm after one year and as dominant in reference areas of the study.

Como & Magni (2009) investigated seasonal variations in benthic communities known to be affected by episodic events of sediment over-enrichment. The authors observed that abundance of *Polydora ciliata* varied seasonally, and suggested this could be a result major accumulation of organic carbon-binding fine sediments in the study site.

Studies by Almeda *et al.* (2009) and Pedersen *et al.* (2010) investigated larval energetic requirements for *Polydora ciliata*, and suggested maximum growth rates were reached at food concentrations ranging from 1.4 to 2.5 µg C/ml depending on larval size, and energetic carbon

requirements of 0.09 to 3.15 µg C l/d, respectively.

Borja *et al.* (2000) and Gittenberger & Van Loon (2011) both assigned *Polydora ciliata* to their AMBI Ecological Group IV 'second-order opportunistic species (slight to pronounced unbalanced situations)', and *Corophium volutator* to Group III 'Species tolerant to excess organic matter enrichment ; these species may occur under normal conditions, but their populations are stimulated by organic enrichment (slight unbalance situations)'.

**Sensitivity assessment.** Connor *et al.* (2004) suggested that as many of the sites featuring this biotope are situated near fish farms, it is possible that it may have developed as the result of moderate nutrient enrichment. Furthermore, the evidence presented suggests that the characterizing species of this biotope are likely to be stimulated by enrichment, and only affected by excessive organic enrichment (above the benchmark level). Resistance and resilience are therefore assessed as **High**, and the biotope considered **Not Sensitive** to organic enrichment involving deposition of 100 gC/m<sup>2</sup>/yr.

## A Physical Pressures

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

<b>Physical change (to another seabed type)</b>	<b>None</b> Q: High A: High C: High	<b>Very Low</b> Q: High A: High C: High	<b>High</b> Q: High A: High C: High
---	--	--	--

SS.SSa.IFiSa.TbAmPo is characterized by the medium to very fine muddy sand substratum which supports the characterizing species. These species have very specific preference of suitable substrata. A change to a rock or artificial substratum would result in the loss of the characterizing species, significantly altering the character of the biotope. The biotope would be lost and/or reclassified.

**Sensitivity assessment.** Resistance to the pressure is considered **None**, and resilience **Very Low** based on the permanent loss of suitable substratum to support the community of the characterizing tube-building polychaete and amphipod species. Sensitivity has been assessed as **High**. Although no specific evidence is described, confidence in this assessment is 'High', due to the incontrovertible nature of this pressure.

<b>Physical change (to another sediment type)</b>	<b>None</b> Q: High A: High C: High	<b>Very Low</b> Q: High A: High C: High	<b>High</b> Q: High A: High C: High
---	--	--	--

The biotope occurs in medium to very fine muddy sand (Connor *et al.*, 2004). A change in sediment type by one Folk Class (based on the Long, 2006 simplification) would result in an increase in the

fraction of sand and gravel in the substratum. The characterizing species would no longer be supported and the biotope would be lost and/or reclassified. For example, *Polydora ciliata* preferably settles on mud; *Corophium* spp. prefers muddier sediments (Watkin, 1941; Flach, 1993); *Scoloplos armiger* prefers sediments 200-350 µm that are enriched with mud (Degraer *et al.*, 2006).

**Sensitivity assessment.** Resistance to the pressure is considered **None**, and resilience **Very Low** based on the loss of suitable substratum to support the community of the characterizing species. Sensitivity has been assessed as **High**.

#### Habitat structure changes - removal of substratum (extraction)

**None**

Q: Medium A: Low C: High

**Medium**

Q: High A: High C: High

**Medium**

Q: Medium A: Low C: High

Removal of the substratum to 30 cm would result in the loss of the mat of *Polydora ciliata* tubes and of *Corophium volutator* that borrows up to 5 cm deep (Meadows & Reid, 1966). Recovery of sediments will be site-specific and will be influenced by currents, wave action and sediment availability (Desprez, 2000). Except in areas of mobile sands, the process tends to be slow (Kenny & Rees, 1996; Desprez, 2000). Boyd *et al.* (2005) found that in a site subject to long-term extraction (25 years), extraction scars were still visible after six years and sediment characteristics were still altered in comparison with reference areas with ongoing effects on the biota. The strongest currents are unable to transport gravel. A further implication of the formation of these depressions is a local drop in current strength associated with the increased water depth, resulting in deposition of finer sediments than those of the surrounding substrate (Desprez, 2000).

**Sensitivity assessment.** Resistance is assessed as **None** as extraction of the sediment will remove the characterizing and associated species present. Resilience is assessed as **Medium** (see resilience section) and sediments may need to recover (where exposed layers are different). Biotope sensitivity is therefore assessed as **Medium**.

#### Abrasion/disturbance of the surface of the substratum or seabed

**Low**

Q: High A: Medium C: High

**High**

Q: High A: High C: High

**Low**

Q: High A: Medium C: High

The tubes of the polychaetes and amphipods are bound only with mucous and are therefore likely to be damaged or removed by abrasion. The soft bodied polychaetes are most likely to suffer mortality, while the more robust amphipods are likely to be more resistant, and mobile enough to avoid impact. The infaunal annelids are predominantly soft bodied, live within a few centimetres of the sediment surface and may expose feeding or respiration structures where they could easily be damaged by a physical disturbance such as a dredge. For example, Ferns *et al.* (2000) reported a decline of 31% in populations of *Scoloplos armiger* (initial density 120/m<sup>2</sup>) in muddy sands and an 83% decline in *Pygospio elegans* (initial density 1850/m<sup>2</sup>) when a mechanical tractor towed harvester was used (in a cockle fishery). *Pygospio elegans* was significantly depleted for >100 days after harvesting (surpassing the study monitoring timeline). *Scoloplos armiger* demonstrated recovery >50 days after harvesting in muddy sands (Ferns *et al.*, 2000).

Additionally, attached epifauna, such as the characterizing *Polydora* community in this biotope, can be entangled and removed by abrasion. Veale *et al.* (2000) reported that the abundance, biomass and production of epifaunal assemblages decreased with increasing fishing effort. Resampling of grounds that were historically studied (from the 1930s) indicates that some upright species have



increased in areas subject to scallop fishing (Bradshaw *et al.*, 2002).

The burrowing life habits of *Corophium volutator* are likely to provide some protection from abrasion at the surface only. However, any abrasion or physical disturbance is likely to reduce the density of *Corophium volutator* by emigration and increased mortality. For example, the sediment turnover caused by cockles and lugworms disturbed the burrows of *Corophium volutator* and caused a significant negative effect on the species density as a result of increased rate of swimming making the amphipod more vulnerable to predation (Flach & De Bruin, 1993, 1994). Furthermore, a number of studies have assessed the effects of trampling on other intertidal amphipods and these assessments are used as a proxy. Ugolini *et al.* (2008) carried out a controlled trampling experiment on *Talitrus saltator*. Plastic cylinders of 110 cm diameter (area 0.95 m<sup>2</sup>) were placed in the sand and all individuals trapped and counted, and 400 steps were made in a cylinder in 15 minutes after the amphipods had reburied. The trampling rate was based on observed number of beach users and therefore represents a realistic level of exposure. Alive individuals were counted at the end of the experiment and 24 hours after. Trampling significantly reduced abundance of the amphipods and after 24 hours the percentage of surviving amphipods dropped to almost zero, while survival rates of control (untrampled) amphipods were unaffected. Abrasion and compaction can therefore kill buried amphipods within sediments.

**Sensitivity assessment.** The evidence presented suggest that erect epifauna and soft bodied individuals, such as those characterizing this biotope are directly exposed to this pressure, which would displace, damage and kill individuals. However, some individuals are likely to survive as individuals can withdraw into burrows. Additionally, although the burrowing life habits of the infaunal community are likely to provide some protection from abrasion at the surface only, physical disturbance is likely to have adverse impacts on the community. Abrasion may also damage the substratum resulting in loss of habitat. Resistance to abrasion is considered **Low** and resilience of the biotope is considered as **High**. The biotope is therefore judged to have **Low** sensitivity to abrasion or disturbance of the surface of the seabed.

#### Penetration or disturbance of the substratum subsurface

None

Q: High A: High C: High

Medium

Q: High A: High C: High

Medium

Q: High A: High C: High

Activities that penetrate below the surface are likely to tear up and remove a significant proportion of the tube building community that characterize this biotope. Additionally, *Corophium volutator* burrows to 5 cm deep and is also likely to be removed. However, in the Columbia river, no significant difference was found in *Corophium volutator* densities before and after dredging a channel and no difference between the dredged site and a control site (McCabe *et al.*, 1998). Presumably, the dredging did cause mortality of *Corophium volutator* but recolonization was so rapid that no difference was found. Additionally, the extraction of cockles by sediment raking and mechanical disturbance and digging for lugworms for bait is likely to cause significant mortality of *Corophium volutator*. Bait digging was found to reduce *Corophium volutator* densities by 39%, juveniles were most affected suffering a 55% reduction in dug areas (Shepherd & Boates, 1999). Bergman & Van Santbrink (2000) found that direct mortality of gammarid amphipods, following a single passage of a beam trawl (in silty sediments where penetration is greater) was 28%. Furthermore, stomach analysis of fish caught scavenging in the tracks of beam trawls found parts of *Ampelisca* spp. indicating that these had been damaged and exposed by the trawl (Kaiser & Spencer, 1994).

Experiments in shallow, wave disturbed areas, using a toothed, clam dredge, found that deposit feeding polychaetes were more impacted than carnivorous species. Dredging resulted in reductions of >90% of *Spiophanes bombyx* immediately post dredging compared with before impact samples and the population reduction persisting for 90 days (although results may be confounded by storm events within the monitoring period which caused sediment mobility). The passage of the dredge across the sediment floor will have killed or injured some organisms that will then be exposed to potential predators/scavengers (Frid *et al.*, 2000; Veale *et al.*, 2000) providing a food source to mobile scavengers including these species. Bergman & Hup (1992) carried out a pre and post-experimental investigation using a 12 m beam trawl. The area was trawled three times over 2 days and samples taken up to 2 weeks after trawling. Some benthic species showed a 10-65% reduction in density after trawling the area three times. There was a significant lowering of densities (40-60%) of polychaete worms, including *Spiophanes bombyx*.

**Sensitivity assessment.** The evidence presented suggest that the community of characterizing species may suffer significant mortality (>75%) as a result of penetrative activities of the seabed. Biotope resistance is therefore assessed as **None** and recovery is assessed as **Medium**. Sensitivity is therefore assessed as **Medium**.

#### Changes in suspended solids (water clarity)

**Low**

Q: Medium A: Medium C: Medium

**High**

Q: High A: High C: High

**Low**

Q: Medium A: Medium C: Medium

The biotope is likely to occur in relatively turbid waters that allow sediment deposition to support the community of characterizing tube-building polychaetes and amphipods, and therefore the species in the biotope are likely to be adapted to turbid conditions. Amphipods are tolerant of high turbidity and gather suspended sediment for the construction of tubes. Mills (1967) reported that feeding by *Ampelisca vadorum* and *Ampelisca abdita* were initiated by the turbidity of the water surrounding the tubes. However, the feeding structures of suspension feeders such as *Ampelisca* spp. may become clogged by large increases in suspended sediment or feeding may be terminated, compromising growth. *Corophium volutator* lives in areas with very high sediment loads and it might be postulated that an increase would not affect them.

Tube-building polychaetes are not likely to be intolerant of high turbidity as they normally inhabit waters with high levels of suspended sediment which they actively fix in the process of tube making. For example, in the Firth of Forth, *Polydora ciliata* formed extensive mats in areas that had an average of 68 mg/l suspended solids and a maximum of approximately 680 mg/l indicating the species is able to tolerate different levels of suspended solids (Read *et al.*, 1982; Read *et al.*, 1983). Daro & Polk (1973) reported that the success of *Polydora* is directly related to the quantities of muds of any origin carried along by rivers or coastal currents. Deposit feeders and tube builders rely on siltation of suspended sediment. A decrease in suspended sediment will reduce this supply and therefore may compromise growth and reproduction.

*Spiophanes bombyx* is found in estuarine regions which experience high levels of turbidity. *Spiophanes bombyx* is a surface deposit feeder and relies on a supply of nutrients at the sediment surface. An increase in turbidity, reducing light availability may reduce primary production by phytoplankton in the water column. Although productivity in the biotope is secondary, a reduction in primary production in the water column may result in reduced food supply to deposit and suspension feeders, which in turn may affect growth rates and fecundity.

**Sensitivity assessment.** An increase in suspended solids at the pressure benchmark level is unlikely to affect the characterizing species of this biotope. However, a decrease in suspended

matter in the biotope could result in limitation of material for tube building activities and also in the substrate no longer being suitable for colonization by new recruits of *Polydora ciliata*, in particular. Resistance of the biotope is therefore considered to be **Low** (loss of 25-75%) and resilience is **High** so the biotope is considered to have **Low** sensitivity to a change in suspended solids at the pressure benchmark level.

### Smothering and siltation rate changes (light)

**Medium**

Q: Medium A: Medium C: High

**High**

Q: High A: High C: High

**Low**

Q: Medium A: Medium C: High

A *Polydora* mud can be up to 50 cm thick, but the animals themselves occupy only the first few centimetres. They either elongate their tubes, or have left them to rebuild close to the surface. Munari & Mistri (2014) investigated the spatio-temporal variation pattern of a benthic community following deposition of dredged material, at a maximum thickness of 30–40 cm. *Polydora ciliata* was amongst the first colonizers of the newly deposited sediments. The authors suggested that it was possible that the individuals migrated vertically through the deep layer of dredged sand. This was based on the results of Roberts *et al.* (1998) who suggested 15 cm as the maximum depth of overburden through which benthic infauna can successfully migrate. After one year, no adverse impact of sand disposal on the benthic fauna was detected on the study site (Munari & Mistri, 2014).

*Corophium volutator* was categorized in AMBI sedimentation Group III – ‘species insensitive to higher amounts of sedimentation, but don’t easily recover from strong fluctuations in sedimentation’ (Gittenberger & Van Loon, 2011). Experimental fences placed on mudflats caused sedimentation rates of 2-2.5 cm/month and reduced *Corophium volutator* densities from approximately 1700 m<sup>-2</sup> to approximately 400 m<sup>-2</sup>. In areas without fences, *Corophium volutator* numbers increased from approximately 1700 per m<sup>-2</sup> to 3500 per m<sup>-2</sup> (Turk & Risk, 1981 cited in Neal & Avant, 2006).

The characterizing polychaetes *Spio filicornis* and *Spiophanes bombyx* were characterized by Gittenberger & Van Loon (2011) in their index of sedimentation tolerance as Group IV species: ‘Although they are sensitive to strong fluctuations in sedimentation, their populations recover relatively quickly and even benefit. This causes their population sizes to increase significantly in areas after a strong fluctuation in sedimentation’ (Gittenberger & Van Loon, 2011). *Pygospio elegans* was classified as ‘Group III’ which do tolerate disturbance and excess organic content by Borja *et al.* (2000).

Where a coarse/impermeable layer was added to the seabed the suitability of the habitat for *Corophium volutator* would be reduced if these could not reach the surface or maintain burrows. Furthermore, a deposition of fine sediment is likely to take a several tidal cycles to clear in the low energetic conditions where this biotope occurs.

**Sensitivity assessment:** Based on the evidence presented, *Polydora ciliata* and other characterizing polychaetes are probably likely to resist and relocate following smothering by 5 cm of sediment. However, the same ‘light’ deposition of fine sediment is likely to cause some mortality of *Corophium* spp.. Resistance is therefore assessed as **Medium** and resilience as **High** and the biotope is considered to have **Low** sensitivity to a ‘light’ deposition of up to 5 cm of fine material in a single discrete event.

**Smothering and siltation rate changes (heavy)****Low**

Q: Medium A: Medium C: High

**High**

Q: High A: High C: High

**Low**

Q: Medium A: Medium C: High

A *Polydora* mud can be up to 50 cm thick, but the animals themselves occupy only the first few centimetres. They either elongate their tubes, or have left them to rebuild close to the surface. Munari & Mistri (2014) investigated the spatio-temporal variation pattern of a benthic community following deposition of dredged material, at a maximum thickness of 30–40 cm. *Polydora ciliata* was amongst the first colonizers of the newly deposited sediments. The authors suggested that it was possible that the individuals migrated vertically through the deep layer of dredged sand. This was based on the results of Roberts *et al.* (1998) who suggested 15 cm as the maximum depth of overburden through which benthic infauna can successfully migrate. After one year, no adverse impact of sand disposal on the benthic fauna was detected on the study site (Munari & Mistri, 2014).

*Corophium volutator* was categorized in AMBI sedimentation Group III – ‘species insensitive to higher amounts of sedimentation, but don’t easily recover from strong fluctuations in sedimentation’ (Gittenberger & Van Loon, 2011). Experimental fences placed on mudflats caused sedimentation rates of 2–2.5 cm/month and reduced *Corophium volutator* densities from approximately 1700 m<sup>-2</sup> to approximately 400 m<sup>-2</sup>. In areas without fences, *Corophium volutator* numbers increased from approximately 1700 per m<sup>2</sup> to 3500 per m<sup>2</sup> (Turk & Risk, 1981 cited in Neal & Avant, 2006). *Pygospio elegans* was classified as ‘Group III’ by Borja *et al.* (2000) as tolerant of disturbance and excess organic content.

Where a coarse/impermeable layer was added to the seabed the suitability of the habitat for *Corophium volutator* would be reduced if these could not reach the surface or maintain burrows. Furthermore, a deposition of fine sediment is likely to take a several tidal cycles to clear in the low energetic conditions where this biotope occurs.

**Sensitivity assessment.** Based on the evidence presented, *Polydora ciliata* and other characterizing polychaetes are probably likely to resist and relocate following smothering by 30 cm of sediment. However, the same ‘heavy’ deposition of fine sediment is likely to result in mortality of *Corophium volutator*. Resistance is therefore assessed as **Low** and resilience as **High** and the biotope is considered to have **Low** sensitivity to a ‘heavy’ deposition of up to 30 cm of fine material in a single discrete event.

**Litter****Not Assessed (NA)**

Q: NR A: NR C: NR

**Not assessed (NA)**

Q: NR A: NR C: NR

**Not assessed (NA)**

Q: NR A: NR C: NR

*Corophium volutator* is widely used in ecotoxicological studies and known to uptake nanoplastics, but toxicity at the current environmental relevant concentrations has yet to be confirmed (Booth *et al.*, 2015). However, *Corophium volutator* forms an important food source for several species of birds and mobile predators such as fish and crabs (Hughes, 1988; Jensen & Kristensen, 1990; Raffaelli *et al.*, 1991; Flach & De Bruin, 1994; Brown *et al.*, 1999), which is likely to result in transition of the particles up the marine food chain. Nevertheless, there was insufficient evidence on which to assess the sensitivity of this biotope to the introduction of litter.

**Electromagnetic changes****No evidence (NEv)**

Q: NR A: NR C: NR

**No evidence (NEv)**

Q: NR A: NR C: NR

**No evidence (NEv)**

Q: NR A: NR C: NR

**No Evidence** was available on which to assess this pressure.

For some amphipods there is evidence for geomagnetic orientation being inhibited or disrupted by the presence of electromagnetic fields or by changing magnetic fields. Arendse & Barendregt (1981) manipulated magnetic fields to alter orientation of the talitrid amphipod *Orchestia cavimana*.

#### Underwater noise changes

Not relevant (NR)

Q: NR A: NR C: NR

Not relevant (NR)

Q: NR A: NR C: NR

Not relevant (NR)

Q: NR A: NR C: NR

*Polydora ciliata* may respond to vibrations from predators or bait diggers by retracting their palps into their tubes. *Corophium volutator* is probably sensitive to surface vibrations but little is known about the effects of noise on invertebrates. However, there is no evidence to suggest that any of the species which characterize the biotope are sensitive to noise or vibration at the level of the benchmark, so the biotope is assessed as **Not Sensitive**.

#### Introduction of light or shading

High

Q: Low A: NR C: NR

High

Q: High A: High C: High

Not sensitive

Q: Low A: NR C: NR

SS.SSa.IFiSa.TbAmPo is a sublittoral biotope (Connor *et al.*, 2004) and therefore not directly dependent on sunlight. However, changes in light in the biotope are likely to affect predation rates and consequently densities of *Corophium volutator*. *Corophium* spp. live in burrows and may not be affected by changes in light although males emerge and crawl over sediments (Meadows, 1964).

**Sensitivity assessment.** Although *Polydora* spp. can perceive light, this pressure is not considered relevant. Additionally, although changes in light may alter predation rates on *Corophium volutator*, it is unlikely to be relevant for the overall composition of the biotope given the species is already likely to be adapted to sustain high levels of predation. The biotope is considered to have **High** resistance and, by default, **High** resilience and therefore is considered **Not Sensitive** to the introduction of light.

#### Barrier to species movement

Not relevant (NR)

Q: NR A: NR C: NR

Not relevant (NR)

Q: NR A: NR C: NR

Not relevant (NR)

Q: NR A: NR C: NR

**Not Relevant** to biotopes restricted to open waters.

#### Death or injury by collision

Not relevant (NR)

Q: NR A: NR C: NR

Not relevant (NR)

Q: NR A: NR C: NR

Not relevant (NR)

Q: NR A: NR C: NR

**Not Relevant** to seabed habitats. NB. Collision by grounding vessels is addressed under surface abrasion.

#### Visual disturbance

High

Q: Low A: NR C: NR

High

Q: High A: High C: High

Not sensitive

Q: Low A: NR C: NR

*Polydora ciliata* exhibits shadow responses and withdraws its palps into its burrow, which is believed to be a defence against predation. The withdrawal of the palps interrupts feeding and

possibly respiration, although the species also shows habituation of the response (Kinne, 1970). *Polydora* is unlikely to be sensitive to visual disturbance caused by passing shipping but may respond to passing divers at close range. Other characterizing polychaetes, such as *Spiophanes bombyx*, also inhabit a tube so visual range is probably very limited. *Corophium volutator* has limited visual acuity and since it spends most of its life in a burrow it is unlikely to be affected by visual disturbances. Furthermore, in substratum choice experiments, *Corophium volutator* preferred substrata independently of whether the dishes were in darkness or illuminated (Meadows, 1964) suggesting visual acuity is low.

**Sensitivity assessment.** Resistance and resilience are assessed as **High** and the biotope judged as **Not Sensitive** to visual disturbance.

## Biological Pressures

	Resistance	Resilience	Sensitivity
<b>Genetic modification &amp; translocation of indigenous species</b>	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR

The important characterizing species in the biotope are not cultivated or likely to be translocated. This pressure is therefore considered **Not Relevant**.

	Resistance	Resilience	Sensitivity
<b>Introduction or spread of invasive non-indigenous species</b>	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR

There is no evidence on the presence of non-indigenous species or impacts of non-indigenous species relevant to this biotope. This pressure is therefore considered **Not Relevant**.

	Resistance	Resilience	Sensitivity
<b>Introduction of microbial pathogens</b>	Low Q: High A: Medium C: High	High Q: High A: High C: High	Low Q: High A: Medium C: High

Introduced organisms (especially parasites or pathogens) are a potential threat in all coastal ecosystems. However, so far, no information was found on microbial pathogens affecting *Polydora ciliata*.

Amphipods may be infected by a number of parasites or pathogens that alter population numbers through changes in host condition, growth, behaviour and reproduction (Green Extabe & Ford, 2014). For example, infection by acanthocephalan larvae may alter behaviour and responses of gammarid amphipods (Bethel & Holmes, 1977). The amphipod *Orchestia gammarellus* is host to the parasitic protist *Marteilia* which has a feminizing effect on populations, with higher ratios of females and intersex males in infected, estuarine populations (Ginsburger-Vogel & Desportes, 1979). *Corophium volutator* is parasitized by several species of trematodes in Europe and North American (McCurdy *et al.*, 2000a; McCurdy *et al.*, 2000b; Mouritsen & Jensen, 1997, cited in Shim *et al.*, 2013). Mass mortalities of *Corophium volutator* have been associated to infestation by trematodes in the Wadden Sea (Jensen & Mouritsen, 1992). A dense field of *Corophium volutator* disappeared completely, and the density of the mud snail *Hydrobia ulvae* declined by 40% during spring 1990 as a result of an epizootic by trematodes. High spring temperature accelerated both

the development rate and the release of infective larval stages of an infectious trematode from the snail *Hydrobia ulvae* (Poulin & Mouritsen, 2006). Studies conducted in the Baltic Sea suggested that increased parasitism by trematode species has a detrimental effect on local amphipods (Meissner & Bick, 1999; Mouritsen & Jensen, 1997 cited in Shim *et al.*, 2013).

**Sensitivity assessment.** Although there are no records of the biotope being affected by the introduction of microbial pathogens in the British Isles, there are reports of mass mortality of characterizing species *Corophium volutator* (Jensen & Mouritsen, 1992). The biotope is therefore at risk and is judged to have **Low** resistance to this pressure. Resilience is assessed as **High** and the biotope is classed as having **Low** sensitivity to the introduction of microbial pathogens.

#### Removal of target species

**High**

Q: High A: Medium C: High

**High**

Q: High A: High C: High

**Not sensitive**

Q: High A: Medium C: High

Fowler (1999) reviewed the effects of bait digging on intertidal fauna, including *Arenicola marina*, which occurs in SS.SSa.IFiSa.TbAmPo lending the biotope sensitive to being targeted by fisheries. Diggers have been reported to remove 50 or 70% of the blow lug population. Heavy commercial exploitation in Budle Bay in winter 1984 removed 4 million worms in 6 weeks, reducing the population from 40 to <1 per m<sup>2</sup>. Recovery occurred within a few months by recolonization from surrounding sediment (Fowler, 1999). However, Cryer *et al.* (1987) reported no recovery for 6 months over summer after mortalities due to bait digging. Mechanical lugworm dredgers have been used in the Dutch Wadden Sea where they removed 17-20 million lugworm/year. A near doubling of the lugworm mortality in dredged areas was reported, resulting in a gradual substantial decline in the local population over a 4 year period. The effects of mechanical lugworm dredging is more severe and can result in the complete removal of *Arenicola marina* (Beukema, 1995; Fowler, 1999). Beukema (1995) noted that the lugworm stock recovered slowly reaching its original level in at least three years. McLusky *et al.* (1983) examined the effects of bait digging on blow lug populations in the Forth Estuary. Dug and infilled areas and unfilled basins left after digging repopulated within 1 month, whereas mounds of dug sediment took showed a reduced population. Basins accumulated fine sediment and organic matter and showed increased population levels for about 2-3 months after digging. Overall, recovery is generally regarded as rapid.

**Sensitivity assessment.** *Arenicola marina* may be targeted by fisheries and the evidence presented suggest that the species abundance can be severely affected. However, *Arenicola marina* is not considered an important characterizing species indicative of sensitivity of SS.SSa.IFiSa.TbAmPo. As the direct, physical impacts are assessed through the abrasion and penetration of the seabed pressures, and this pressure considers only the ecological or biological effects of removal of the targeted species, it is considered that removal of *Arenicola marina* is unlikely to have an impact on the character of the biotope. Furthermore, the biotope occurs in the shallow sublittoral and is unlikely to be exposed at low tide and hence targeted by bait diggers. Resistance and resilience are therefore assessed as **High**, and the biotope considered **Not Sensitive** to this pressure.

#### Removal of non-target species

**Low**

Q: High A: High C: High

**High**

Q: High A: High C: High

**Low**

Q: High A: High C: High

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. The characterizing species in this biotope are highly likely to be damaged or directly removed by static or mobile gears that are targeting other species (see abrasion and penetration pressures).

**Sensitivity assessment.** Loss of the characterizing species of this biotope is likely to occur as a result of unintentional removal. Removal of the characterizing species would result in the biotope being lost. Thus, the biotope is considered to have a resistance of **Low** to this pressure and to have **High** resilience, resulting in the sensitivity being judged as **Low**.



## Bibliography

- Aberkali, H.B. & Trueman, E.R., 1985. Effects of environmental stress on marine bivalve molluscs. *Advances in Marine Biology*, **22**, 101-198.
- Almeda, R., Pedersen, T.M., Jakobsen, H.H., Alcaraz, M., Calbet, A. & Hansen, B.W., 2009. Feeding and growth kinetics of the planktotrophic larvae of the spionid polychaete *Polydora ciliata* (Johnston). *Journal of Experimental Marine Biology and Ecology*, **382** (1), 61-68.
- Anger V., 1984. Reproduction in *Pygospio-elegans* Spionidae in relation to its geographical origin and to environmental conditions a preliminary report. Fischer, A. and H.-D. Pfannenstiel, Fortschritte der Zoologie. pp. 45-52.
- Arendse, M.C. & Barendregt, A., 1981. Magnetic orientation in the semi-terrestrial amphipod, *Orchestia cavimana*, and its interrelationship with photo-orientation and water loss. *Physiological Entomology*, **6** (4), 333-342.
- Bailey-Brook, J.H., 1976. Habitats of tubicolous polychaetes from the Hawaiian Islands and Johnston Atoll. *Pacific Science*, **30**, 69-81.
- Barnes, R.S.K. & Hughes, R.N., 1992. *An introduction to marine ecology*. Oxford: Blackwell Scientific Publications.
- Barnes, R.S.K., 1994. *The brackish-water fauna of northwestern Europe*. Cambridge: Cambridge University Press.
- Bat, L., Raffaelli, D. & Marr, I.L., 1998. The accumulation of copper, zinc and cadmium by the amphipod *Corophium volutator* (Pallas). *Journal of Experimental Marine Biology and Ecology*, **223**, 167-184.
- Beaumont, A.R., Newman, P.B., Mills, D.K., Waldock, M.J., Miller, D. & Waite, M.E., 1989. Sandy-substrate microcosm studies on tributyl tin (TBT) toxicity to marine organisms. *Scientia Marina*, **53**, 737-743.
- Bergman, M.J.N. & Hup, M., 1992. Direct effects of beam trawling on macro-fauna in a sandy sediment in the southern North Sea. *ICES Journal of Marine Science*, **49**, 5-11.
- Bergman, M.J.N. & Van Santbrink, J.W., 2000b. Fishing mortality of populations of megafauna in sandy sediments. In *The effects of fishing on non-target species and habitats* (ed. M.J. Kaiser & S.J. de Groot), 49-68. Oxford: Blackwell Science.
- Bethel, W.M. & Holmes, J.C., 1977. Increased vulnerability of amphipods to predation owing to altered behavior induced by larval acanthocephalans. *Canadian Journal of Zoology*, **55** (1), 110-115.
- Beukema, J.J. & De Vlas, J., 1979. Population parameters of the lugworm, *Arenicola marina*, living on tidal flats in the Dutch Wadden Sea. *Netherlands Journal of Sea Research*, **13**, 331-353.
- Beukema, J.J., 1995. Long-term effects of mechanical harvesting of lugworms *Arenicola marina* on the zoobenthic community of a tidal flat in the Wadden Sea. *Netherlands Journal of Sea Research*, **33**, 219-227.
- Black, K.D., Fleming, S. Nickell, T.D. & Pereira, P.M.F. 1997. The effects of ivermectin, used to control sea lice on caged farmed salmonids, on infaunal polychaetes. *ICES Journal of Marine Science*, **54**, 276-279.
- Blackstock, J. & Barnes, M., 1982. The Loch Eil project: biochemical composition of the polychaete, *Glycera alba* (Müller), from Loch Eil. *Journal of Experimental Marine Biology and Ecology*, **57** (1), 85-92.
- Blanchard, M., 1997. Spread of the slipper limpet *Crepidula fornicata* (L.1758) in Europe. Current state and consequences. *Scientia Marina*, **61**, Supplement 9, 109-118.
- Bohn, K., Richardson, C.A. & Jenkins, S.R., 2015. The distribution of the invasive non-native gastropod *Crepidula fornicata* in the Milford Haven Waterway, its northernmost population along the west coast of Britain. *Helgoland Marine Research*, **69** (4), 313.
- Boon, J.P., Zantvoort, M.B., Govaert, M.J.M.A. & Duinker, J.C., 1985. Organochlorines in benthic polychaetes (*Nephtys* spp.) and sediments from the southern North Sea. Identification of individual PCB components. *Netherlands Journal of Sea Research*, **19**, 93-109.
- Booth, A. M., Hansen, B.H., Frenzel, M., Johnsen, H. & Altin, D., 2015. Uptake and toxicity of methylmethacrylate-based nanoplastic particles in aquatic organisms. *Environmental Toxicology and Chemistry*, **9999**, 1-9.
- Borja, A., Franco, J. & Perez, V., 2000. A marine biotic index to establish the ecological quality of soft-bottom benthos within European estuarine and coastal environments. *Marine Pollution Bulletin*, **40** (12), 1100-1114.
- Boulcott, P. & Howell, T.R.W., 2011. The impact of scallop dredging on rocky-reef substrata. *Fisheries Research* (Amsterdam), **110** (3), 415-420.
- Bousfield, E.L., 1973. *Shallow-water gammaridean Amphipoda of New England*. London: Cornell University Press.
- Boyd, S., Limpenny, D., Rees, H. & Cooper, K., 2005. The effects of marine sand and gravel extraction on the macrobenthos at a commercial dredging site (results 6 years post-dredging). *ICES Journal of Marine Science: Journal du Conseil*, **62** (2), 145-162.
- Bradshaw, C., Veale, L.O., Hill, A.S. & Brand, A.R., 2002. The role of scallop-dredge disturbance in long-term changes in Irish Sea benthic communities: a re-analysis of an historical dataset. *Journal of Sea Research*, **47**, 161-184.
- Brils, J.M., Huwer, S.L., Kater, B.J., Schout, P.G., Harmsen, J., Delvigne, G.A.L. & Scholten, M.C.T., 2002. Oil effect in freshly spiked marine sediment on *Vibrio fischeri*, *Corophium volutator*, and *Echinocardium caudatum*. *Environmental Toxicology and Chemistry*, **21**, 2242-2251.
- Brown, R.J., Conradi, M. & Depledge, M.H., 1999. Long-term exposure to 4-nonylphenol affects sexual differentiation and growth of the amphipod *Corophium volutator* (Pallas, 1766). *Science of the Total Environment*, **233**, 77-88.

- Bryan, G.W. & Gibbs, P.E., 1983. *Heavy metals from the Fal estuary, Cornwall: a study of long-term contamination by mining waste and its effects on estuarine organisms*. Plymouth: Marine Biological Association of the United Kingdom. [Occasional Publication, no. 2.]
- Bryan, G.W., 1984. Pollution due to heavy metals and their compounds. In *Marine Ecology: A Comprehensive, Integrated Treatise on Life in the Oceans and Coastal Waters*, vol. 5. *Ocean Management*, part 3, (ed. O. Kinne), pp.1289-1431. New York: John Wiley & Sons.
- Bryant, V., McLusky, D.S., Roddie, K. & Newbery, D.M., 1984. Effect of temperature and salinity on the toxicity of chromium to three estuarine invertebrates (*Corophium volutator*, *Macoma balthica*, *Nereis diversicolor*). *Marine Ecology Progress Series*, **20**, 137-149.
- Bryant, V., Newbery, D.M., McLusky, D.S. & Campbell, R., 1985. Effect of temperature and salinity on the toxicity of arsenic to three estuarine invertebrates (*Corophium volutator*, *Macoma balthica*, *Tubifex costatus*). *Marine Ecology Progress Series*, **24**, 129-137.
- Bryant, V., Newbery, D.M., McLusky, D.S. & Campbell, R., 1985a. Effect of temperature and salinity on the toxicity of nickel and zinc to two estuarine invertebrates (*Corophium volutator*, *Macoma balthica*). *Marine Ecology Progress Series*, **24**, 139-153.
- Callier, M. D., McKindsey, C.W. & Desrosiers, G., 2007. Multi-scale spatial variations in benthic sediment geochemistry and macrofaunal communities under a suspended mussel culture. *Marine Ecology Progress Series*, **348**, 103-115.
- Capasso, E., Jenkins, S., Frost, M. & Hinz, H., 2010. Investigation of benthic community change over a century-wide scale in the western English Channel. *Journal of the Marine Biological Association of the United Kingdom*, **90** (06), 1161-1172.
- Cole, S., Codling, I.D., Parr, W. & Zabel, T., 1999. Guidelines for managing water quality impacts within UK European Marine sites. *Natura 2000 report prepared for the UK Marine SACs Project*. 441 pp., Swindon: Water Research Council on behalf of EN, SNH, CCW, JNCC, SAMS and EHS. [UK Marine SACs Project.], <http://www.ukmarinesac.org.uk/>
- Collie, J.S., Escanero, G.A. & Valentine, P.C., 1997. Effects of bottom fishing on the benthic megafauna of Georges Bank. *Marine Ecology Progress Series*, **155**, 159-172.
- Collie, J.S., Hall, S.J., Kaiser, M.J. & Poiner, I.R., 2000. A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, **69** (5), 785-798.
- Collier, L.M. & Pinn, E.H., 1998. An assessment of the acute impact of the sea lice treatment Ivermectin on a benthic community. *Journal of Experimental Marine Biology and Ecology*, **230**, 131-147.
- Como, S. & Magni, P., 2009. Temporal changes of a macrobenthic assemblage in harsh lagoon sediments. *Estuarine, Coastal and Shelf Science*, **83** (4), 638-646.
- Conan, G., 1982. The long-term effects of the Amoco Cadiz oil spill. *Philosophical Transactions of the Royal Society of London B*, **297**, 323-333.
- Connor, D.W., Allen, J.H., Golding, N., Howell, K.L., Lieberknecht, L.M., Northen, K.O. & Reker, J.B., 2004. The Marine Habitat Classification for Britain and Ireland. Version 04.05. ISBN 1 861 07561 8. In JNCC (2015), *The Marine Habitat Classification for Britain and Ireland Version 15.03*. [2019-07-24]. Joint Nature Conservation Committee, Peterborough. Available from <https://mhc.jncc.gov.uk/>
- Connor, D.W., Dalkin, M.J., Hill, T.O., Holt, R.H.F. & Sanderson, W.G., 1997a. Marine biotope classification for Britain and Ireland. Vol. 2. Sublittoral biotopes. *Joint Nature Conservation Committee, Peterborough, JNCC Report no. 230, Version 97.06., Joint Nature Conservation Committee, Peterborough, JNCC Report no. 230, Version 97.06.*
- Conradi, M. & Depledge, M.H., 1999. Effects of zinc on the life-cycle, growth and reproduction of the marine amphipod *Corophium volutator*. *Marine Ecology Progress Series*, **176**, 131-138.
- Cooper, K., Ware, S., Vanstaen, K. & Barry, J., 2011. Gravel seeding - A suitable technique for restoring the seabed following marine aggregate dredging? *Estuarine, Coastal and Shelf Science*, **91** (1), 121-132.
- Crisp, D.J. (ed.), 1964. The effects of the severe winter of 1962-63 on marine life in Britain. *Journal of Animal Ecology*, **33**, 165-210.
- Cryer, M., Whittle, B.N. & Williams, K., 1987. The impact of bait collection by anglers on marine intertidal invertebrates. *Biological Conservation*, **42**, 83-93.
- Daro, M.H. & Polk, P., 1973. The autecology of *Polydora ciliata* along the Belgian coast. *Netherlands Journal of Sea Research*, **6**, 130-140.
- Dauvin, J-C. & Bellan-Santini, D., 1990. An overview of the amphipod genus *Haploops* (Ampeliscidae). *Journal of the Marine Biological Association of the United Kingdom*, **70**, 887-903.
- Dauvin, J.C., 1988b. Biologie, dynamique, et production de populations de Crustacés amphipodes de la Manche occidentale. 1. *Ampelisca tenuicornis* Liljeborg. *Journal of Experimental Marine Biology and Ecology*, **118**, 55-84.
- Dauvin, J.C., 1988c. The life cycle, population dynamics and production of the populations of amphipod crustaceans of the English Channel. 3. *Ampelisca typica* (Bate). *Journal of Experimental Marine Biology and Ecology*, **121**, 1-22.
- Dauvin, J.C., 1988d. Life cycle, dynamics, and productivity of Crustacea-Amphipoda from the western English Channel. 4. *Ampelisca armoricana* Bellan-Santini et Dauvin. *Journal of Experimental Marine Biology and Ecology*, **123**, 235-252
- Dauvin, J.C., 1988e. Biologie, dynamique, et production de populations de crustacés amphipodes de la Manche occidentale. 2. *Ampelisca brevicornis* (Costa). *Journal of Experimental Marine Biology and Ecology*, **119**, 213-233.
- Dauvin, J.C., 1989. Life cycle, dynamics and productivity of Crustacea-Amphipoda from the western English Channel. 5. *Ampelisca sarsi* Chevreux. *Journal of Experimental Marine Biology and Ecology*, **128**, 31-56.
- Dauvin, J.C., 1998. The fine sand *Abra alba* community of the Bay of Morlaix twenty years after the Amoco Cadiz oil spill. *Marine*

*Pollution Bulletin*, **36**, 669-676.

Davies, C.E. & Moss, D., 1998. European Union Nature Information System (EUNIS) Habitat Classification. *Report to European Topic Centre on Nature Conservation from the Institute of Terrestrial Ecology, Monks Wood, Cambridgeshire*. [Final draft with further revisions to marine habitats.], Brussels: European Environment Agency.

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

de-la-Ossa-Carretero, J., Del-Pilar-Ruso, Y., Loya-Fernández, A., Ferrero-Vicente, L., Marco-Méndez, C., Martínez-García, E. & Sánchez-Lizaso, J., 2016. Response of amphipod assemblages to desalination brine discharge: impact and recovery. *Estuarine, Coastal and Shelf Science*, **172**, 13-23

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

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

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

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

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

Dorsett, D.A., 1961. The reproduction and maintenance of *Polydora ciliata* (Johnst.) at Whitstable. *Journal of the Marine Biological Association of the United Kingdom*, **41**, 383-396.

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

Eagle, R.A., 1975. Natural fluctuations in a soft bottom benthic community. *Journal of the Marine Biological Association of the United Kingdom*, **55**, 865-878.

Eisler, R., 1977. Toxicity evaluation of a complex meta mixture to the softshell clam *Mya arenaria*. *Marine Biology*, **43**, 265-276.

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

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

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

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

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

Fish, J.D. & Fish, S., 1974. The breeding cycle and growth of *Hydrobia ulvae* in the Dovey estuary. *Journal of the Marine Biological Association of the United Kingdom*, **54**, 685-697.

Fish, J.D. & Fish, S., 1996. *A student's guide to the seashore*. Cambridge: Cambridge University Press.

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

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

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

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

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

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

Ford, R.B. & Paterson, D.M., 2001. Behaviour of *Corophium volutator* in still versus flowing water. *Estuarine, Coastal and Shelf Science*, **52**, 357-362.

Fowler, S.L., 1999. Guidelines for managing the collection of bait and other shoreline animals within UK European marine sites. *Natura 2000 report prepared by the Nature Conservation Bureau Ltd. for the UK Marine SACs Project*, 132 pp., Peterborough: English

- Nature (UK Marine SACs Project)., <http://www.english-nature.org.uk/uk-marine/reports/reports.htm>
- Fretter, V. & Graham, A., 1981. The Prosobranch Molluscs of Britain and Denmark. Part 6. molluscs of Britain and Denmark. part 6. *Journal of Molluscan Studies*, Supplement 9, 309-313.
- Frid, C.L., Harwood, K.G., Hall, S.J. & Hall, J.A., 2000. Long-term changes in the benthic communities on North Sea fishing grounds. *ICES Journal of Marine Science*, **57** (5), 1303.
- Gamble, J., 1970. Anaerobic survival of the crustaceans *Corophium volutator*, *C. arenarium* and *Tanais chevreuxi*. *Journal of the Marine Biological Association of the United Kingdom*, **50** (03), 657-671.
- Gamenick, I., Jahn, A., Vopel, K. & Giere, O., 1996. Hypoxia and sulphide as structuring factors in a macrozoobenthic community on the Baltic Sea shore: Colonization studies and tolerance experiments. *Marine Ecology Progress Series*, **144**, 73-85.
- Gameson, 1982. The quality of the Humber Estuary, 1961-1981, *Yorkshire Water Authority*.
- Gerdol, V. & Hughes, R.G., 1993. Effect of the amphipod *Corophium volutator* on the colonisation of mud by the halophyte *Salicornia europea*. *Marine Ecology Progress Series*, **97**, 61-69.
- Gibson, G.D. & Harvey, J., 2000. Morphogenesis during asexual reproduction in *Pygospio elegans* Claparede (Annelida, Polychaeta). *The Biological Bulletin*, **199** (1), 41-49.
- Gilkinson, K., Paulin, M., Hurley, S. & Schwinghamer, P., 1998. Impacts of trawl door scouring on infaunal bivalves: results of a physical trawl door model/dense sand interaction. *Journal of Experimental Marine Biology and Ecology*, **224** (2), 291-312.
- Gilkinson, K.D., Gordon, D.C., MacIsaac, K.G., McKeown, D.L., Kenchington, E.L., Bourbonnais, C. & Vass, W.P., 2005. Immediate impacts and recovery trajectories of macrofaunal communities following hydraulic clam dredging on Banquereau, eastern Canada. *ICES Journal of Marine Science: Journal du Conseil*, **62** (5), 925-947.
- Ginsburger-Vogel, T. & Desportes, I., 1979. Structure and biology of *Marteilia* sp. in the amphipod *Orchestia gammarellus*. *Marine Fisheries Review*, **41**, 3-7.
- Gittenberger, A. & Van Loon, W.M.G.M., 2011. Common Marine Macrozoobenthos Species in the Netherlands, their Characteristics and Sensitivities to Environmental Pressures. GiMaRIS report no 2011.08. DOI: [10.13140/RG.2.1.3135.7521](https://doi.org/10.13140/RG.2.1.3135.7521)
- Gogina, M., Glockzin, M. & Zettler, M.L., 2010. Distribution of benthic macrofaunal communities in the western Baltic Sea with regard to near-bottom environmental parameters. 2. Modelling and prediction. *Journal of Marine Systems*, **80**, 57-70.
- Grassle, J.F. & Grassle, J.P., 1974. Opportunistic life histories and genetic systems in marine benthic polychaetes. *Journal of Marine Research*, **32**, 253-284.
- Gray, J.S., Clarke, K.R., Warwick, R.M. & Hobbs, G., 1990. Detection of initial effects of pollution on marine benthos - an example from the Ekofisk and Eldfisk oilfields, North Sea. *Marine Ecology Progress Series*, **66** (3), 285-299.
- Green Etxabe, A. & Ford, A., 2014. Do demon shrimp carry demon parasites? *Freshwater Biological Association News*, **62**, 10-11.
- Green, J., 1961. A biology of Crustacea. London: H.F. & G. Witherby Ltd. 180pp.
- Green, N.W., 1983. Key colonisation strategies in a pollution-perturbed environment. In *Fluctuations and Succession in Marine Ecosystems: Proceedings of the 17th European Symposium on Marine Biology, Brest, France, 27 September - 1st October 1982*. *Oceanologica Acta*, 93-97.
- Gudmundsson, H., 1985. Life history patterns of polychaete species of the family spionidae. *Journal of the Marine Biological Association of the United Kingdom*, **65**, 93-111.
- Gulliksen, B., 1977. Studies from the U.W.L. "Helgoland" on the macrobenthic fauna of rocks and boulders in Lübeck Bay (western Baltic Sea). *Helgoländer wissenschaftliche Meeresunters*, **30**, 519-526.
- Hall, S.J., 1994. Physical disturbance and marine benthic communities: life in unconsolidated sediments. *Oceanography and Marine Biology: an Annual Review*, **32**, 179-239.
- Hansen, B. W., Stenalt, E., Petersen, J.K. & Ellegaard, C., 2002. Invertebrate re-colonisation in Mariager Fjord (Denmark) after severe hypoxia. I. Zooplankton and settlement. *Ophelia* **56** (3), 197-213.
- Harms, J. & Anger, K., 1983. Seasonal, annual, and spatial variation in the development of hard bottom communities. *Helgoländer Meeresuntersuchungen*, **36**, 137-150.
- Harris, G.J. & Morgan, E., 1984a. The effects of salinity changes on the endogenous circa-tidal rhythm of the amphipod *Corophium volutator* (Pallas). *Marine Behaviour and Physiology*, **10**, 199-217.
- Harris, G.J. & Morgan, E., 1984b. The effects of ethanol, valinomycin and cycloheximide on the endogenous circa-tidal rhythm of the estuarine amphipod *Corophium volutator* (Pallas). *Marine Behaviour and Physiology*, **10**, 219-233.
- Hayward, P.J. 1994. *Animals of sandy shores*. Slough, England: The Richmond Publishing Co. Ltd. [Naturalists' Handbook 21.]
- Hayward, P.J. & Ryland, J.S. (ed.) 1995b. *Handbook of the marine fauna of North-West Europe*. Oxford: Oxford University Press.
- Hiscock, K., 1983. Water movement. In *Sublittoral ecology. The ecology of shallow sublittoral benthos* (ed. R. Earll & D.G. Erwin), pp. 58-96. Oxford: Clarendon Press.
- Hiscock, K., Langmead, O., Warwick, R. & Smith, A., 2005a. Identification of seabed indicator species to support implementation of the EU Habitats and Water Framework Directives. *Report to the Joint Nature Conservation Committee and the Environment Agency* The Marine Biological Association, Plymouth, 77 pp.
- Hjulström, F., 1939. Transportation of detritus by moving water: Part 1. Transportation. *Recent Marine Sediments, a Symposium*

(ed. P.D. Trask), pp. 5-31. Dover Publications, Inc.

Hoare, R. & Hiscock, K., 1974. An ecological survey of the rocky coast adjacent to the effluent of a bromine extraction plant. *Estuarine and Coastal Marine Science*, **2** (4), 329-348.

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

Hong, J. & Reish, D.J., 1987. Acute toxicity of cadmium to eight species of marine amphipod and isopod crustaceans from southern California. *Bulletin of Environmental Contamination and Toxicology*, **39**, 884-888.

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

Huthnance, J., 2010. Ocean Processes Feeder Report. London, *DEFRA on behalf of the United Kingdom Marine Monitoring and Assessment Strategy (UKMMAS) Community*.

Jacobs, R.P.W.M., 1980. Effects of the *Amoco Cadiz* oil spill on the seagrass community at Roscoff with special reference to the benthic infauna. *Marine Ecology Progress Series*, **2**, 207-212.

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

Jensen, K.T. & Mouritsen K.N., 1992. Mass mortality in two common soft bottom invertebrates, *Hydrobia ulvae* and *Corophium volutator*, the possible role of trematodes. *Helgolander Meeresuntersuchungen*, **46**, 329-339.

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

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

JNCC (Joint Nature Conservation Committee), 1999. *Marine Environment Resource Mapping And Information Database (MERMAID): Marine Nature Conservation Review Survey Database*. [on-line] <http://www.jncc.gov.uk/mermaid>

Kaiser, M.J. & Spencer, B.E., 1994. Fish scavenging behaviour in recently trawled areas. *Marine Ecology Progress Series*, **112** (1-2), 41-49.

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

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

Kesaniemi, J.E., Geuverink, E. & Knott, K.E., 2012. Polymorphism in developmental mode and its effect on population genetic structure of a Spionid Polychaete, *Pygospio elegans*. *Integrative and Comparative Biology*, **52** (1), 181-196.

Kinne, O. (ed.), 1970. *Marine Ecology: A Comprehensive Treatise on Life in Oceans and Coastal Waters. Vol. 1 Environmental Factors Part 1*. Chichester: John Wiley & Sons

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

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

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

Kruse, I., Strasser, M. & Thiermann, F., 2004. The role of ecological divergence in speciation between intertidal and subtidal *Scoloplos armiger* (Polychaeta, Orbiniidae). *Journal of Sea Research*, **51**, 53-62.

Lagadeuc, Y., 1991. Mud substrate produced by *Polydora ciliata* (Johnston, 1828) (Polychaeta, Annelida) - origin and influence on fixation of larvae. *Cahiers de Biologie Marine*, **32**, 439-450.

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

Le Bris, H. & Glemarec, M., 1995. Macrobenthic communities of oxygen under-saturated ecosystems: The Bay of Vilaine, southern Brittany. *Oceanologica Acta*, **18**, 573-581.

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

Long, D., 2006. BGS detailed explanation of seabed sediment modified Folk classification. Available from: [http://www.emodnet-seabedhabitats.eu/PDF/GMHM3\\_Detailed\\_explanation\\_of\\_seabed\\_sediment\\_classification.pdf](http://www.emodnet-seabedhabitats.eu/PDF/GMHM3_Detailed_explanation_of_seabed_sediment_classification.pdf)

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

Maurer, D. & Lethem, W., 1980. Dominant species of polychaetous annelids of Georges Bank. *Marine Ecology Progress Series*, **3**,

135-144.

Maurer, D., Keck, R.T., Tinsman, J.C. & Leathem, W.A., 1982. Vertical migration and mortality of benthos in dredged material: Part III—polychaeta. *Marine Environmental Research*, **6** (1), 49-68.

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

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

McCurdy, D.G., Boates, J.S. & Forbes, M.R., 2000. Reproductive synchrony in the intertidal amphipod *Corophium volutator*. *Oikos*, **88**, 301-308.

McDermott, J.J., 1984. The feeding biology of *Nipponnemertes pulcher* (Johnston) (Hoploneuridae), with some ecological implications. *Ophelia*, **23**, 1-21.

McLusky, D., Anderson, F. & Wolfe-Murphy, S., 1983. Distribution and population recovery of *Arenicola marina* and other benthic fauna after bait digging. *Marine Ecology Progress Series*, **11** (2), 173-179.

McLusky, D.S., 1967. Some effects of salinity on the survival, moulting, and growth of *Corophium volutator* (Amphipoda). *Journal of the Marine Biological Association of the United Kingdom*, **47**, 607-617.

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

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

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

Meador, J.P., Varanasi, U. & Krone, C.A., 1993. Differential sensitivity of marine infaunal amphipods to tributyltin. *Marine Biology*, **116**, 231-239.

Meadows, P., 1964. Substrate selection by *Corophium* species: the particle size of substrates. *The Journal of Animal Ecology*, **33**, 387-394.

Meadows, P. & Reid, A. (1966). The behaviour of *Corophium volutator* (Crustacea: Amphipoda). *Journal of Zoology* **150**(4): 387-399

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

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

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

MES, 2010. *Marine Macrofauna Genus Trait Handbook*. Marine Ecological Surveys Limited. <http://www.genustrait handbook.org.uk/>

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

Mills, E.L., 1967. The biology of an ampeliscid amphipod crustacean sibling species pair. *Journal of the Fisheries Research Board of Canada*, **24**, 305-355.

Miramand, P., Germain, P. & Camus, H., 1982. Uptake of americium and plutonium from contaminated sediments by three benthic species: *Arenicola marina*, *Corophium volutator* and *Scrobicularia plana*. *Marine Ecology Progress Series*, **7**, 59-65.

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

Mouritsen, K. N., Mouritsen, L.T. & Jensen, K.T., 1998. Change of topography and sediment characteristics on an intertidal mudflat following mass-mortality of the amphipod *Corophium volutator*. *Journal of the Marine Biological Association of the United Kingdom*, **78** (4), 1167-1180.

Mouritsen, K.N., Tompkins, D.M. & Poulin, R., 2005. Climate warming may cause a parasite-induced collapse in coastal amphipod populations. *Oecologia*, **146**, 476-483.

Munari, C. & Mistri, M., 2014. Spatio-temporal pattern of community development in dredged material used for habitat enhancement: A study case in a brackish lagoon. *Marine Pollution Bulletin* **89** (1-2), 340-347.

Murina, V., 1997. Pelagic larvae of Black Sea Polychaeta. *Bulletin of Marine Science*, **60**, 427-432.

Mustaquim, J., 1986. Morphological variation in *Polydora ciliata* complex (Polychaeta, Annelida). *Zoological Journal of the Linnean Society*, **86**, 75-88.

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

Neves de Carvalho, A., Vaz, A.S.L., Sérgio, T.I.B. & Santos, P.J.T.d., 2013. Sustainability of bait fishing harvesting in estuarine ecosystems: Case study in the Local Natural Reserve of Douro Estuary, Portugal estuarinos: Caso de estudo na Reserva Natural

- Local do Estuário do Douro, Portugal. *Revista de Gestão Costeira Integrada*, **13** (2), 157-168.
- Newell, R., Seiderer, L. & Hitchcock, D., 1998. The impact of dredging works in coastal waters: a review of the sensitivity to disturbance and subsequent recovery of biological resources on the sea bed. *Oceanography and Marine Biology: An Annual Review*, **36**, 127-178.
- Nichols, F.H. & Thompson, J.K., 1985. Persistence of an introduced mudflat community in South San Francisco Bay, California. *Marine Ecology Progress Series*, **24**, 83-97.
- Niermann, U., Bauerfeind, E., Hickel, W. & Westernhagen, H.V., 1990. The recovery of benthos following the impact of low oxygen content in the German Bight. *Netherlands Journal of Sea Research*, **25**, 215-226.
- OBIS, 2016. Ocean Biogeographic Information System (OBIS). <http://www.iobis.org>, 2016-03-15
- Olafsson, E.B. & Persson, L.E., 1986. The interaction between *Nereis diversicolor* (Muller) and *Corophium volutator* (Pallas) as a structuring force in a shallow brackish sediment. *Journal of Experimental Marine Biology and Ecology*, **103**, 103-117.
- OSPAR, 2000. OSPAR decision 2000/3 on the use of organic-phase drilling fluids (OPF) and the discharge of OPF-contaminated cuttings. Summary Record OSPAR 2000. OSPAR 00/20/1-E, Annex 18. Copenhagen, 26-30 June.
- Pardal, M.A., Marques, J.-C. & Bellan, G., 1993. Spatial distribution and seasonal variation of subtidal polychaete populations in the Mondego estuary (western Portugal). *Cahiers de Biologie Marine*, **34**, 497-512.
- Pearson, T.H. & Rosenberg, R., 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. *Oceanography and Marine Biology: an Annual Review*, **16**, 229-311.
- Pedersen, T. M., Almeda, R., Fotel, F.L., Jakobsen, Hans H., Mariani, P. & Hansen, B.W., 2010. Larval growth in the dominant polychaete *Polydora ciliata* is food-limited in a eutrophic Danish estuary (Isefjord). *Marine Ecology Progress Series*, **407**, 99-110.
- Picton, B.E. & Costello, M.J., 1998. *BioMar* biotope viewer: a guide to marine habitats, fauna and flora of Britain and Ireland. [CD-ROM] Environmental Sciences Unit, Trinity College, Dublin.
- Poggiale, J.C. & Dauvin, J.C., 2001. Long term dynamics of three benthic *Ampelisca* (Crustacea - Amphipoda) populations from the Bay of Morlaix (western English Channel) related to their disappearance after the *Amoco Cadiz* oil spill. *Marine Ecology Progress Series*, **214**, 201-209.
- Poulin, R. & Mouritsen, K.N., 2006. Climate change, parasitism and the structure of intertidal ecosystems. *Journal of Helminthology*, **80** (2), 183-192.
- Powell, C.E., 1979. Isopods other than cyathura (Arthropoda: Crustacea: Isopoda). In *Pollution ecology of estuarine invertebrates* (ed. C.W. Hart & S.L.H. Fuller), 325-338. New York: Academic Press.
- Powilleit, M., Graf, G., Kleine, J., Riethmuller, R., Stockmann, K., Wetzel, M.A. & Koop, J.H.E., 2009. Experiments on the survival of six brackish macro-invertebrates from the Baltic Sea after dredged spoil coverage and its implications for the field. *Journal of Marine Systems*, **75** (3-4), 441-451.
- Raffaelli, D., Limia, J., Hull, S. & Pont, S., 1991. Interactions between the amphipod *Corophium volutator* and macroalgal mats on estuarine mudflats. *Journal of the Marine Biological Association of the United Kingdom*, **71**, 899-908.
- Read, P.A., Anderson, K.J., Matthews, J.E., Watson, P.G., Halliday, M.C. & Shiells, G.M., 1982. Water quality in the Firth of Forth. *Marine Pollution Bulletin*, **13**, 421-425.
- Read, P.A., Anderson, K.J., Matthews, J.E., Watson, P.G., Halliday, M.C. & Shiells, G.M., 1983. Effects of pollution on the benthos of the Firth of Forth. *Marine Pollution Bulletin*, **14**, 12-16.
- Rees, E., Nicolaidou, A. & Laskaridou, P., 1976. The effects of storms on the dynamics of shallow water benthic associations. In *Proceedings of the 11th European Symposium on Marine Biology, Galway, 5-11 October, 1976. Biology of benthic organisms* (ed. B.F., Keegan; P., O'Ceidigh & P.J.S., Boaden), pp. 465-474.
- Reish, D.J., 1979. Bristle Worms (Annelida: Polychaeta) In *Pollution Ecology of Estuarine Invertebrates*, (eds. Hart, C.W. & Fuller, S.L.H.), 78-118. Academic Press Inc, New York.
- Rhoads, D.C. & Young, D.K., 1970. The influence of deposit-feeding organisms on sediment stability and community trophic structure. *Journal of Marine Research*, **28**, 150-178.
- Riedel, B., Zuschin, M. & Stachowitsch, M., 2012. Tolerance of benthic macrofauna to hypoxia and anoxia in shallow coastal seas: a realistic scenario. *Marine Ecology Progress Series*, **458**, 39-52.
- Riera, R., Tuya, F., Ramos, E., Rodríguez, M. & Monterroso, Ó., 2012. Variability of macrofaunal assemblages on the surroundings of a brine disposal. *Desalination*, **291**, 94-100.
- Roberts, R. D., Gregory, M.R. & Foster, B.A., 1998. Developing an efficient macrofauna monitoring index from an impact study—a dredge spoil example. *Marine Pollution Bulletin*, **36** (3), 231-235.
- Roche, C., Lyons, D.O., O'Connor, B. 2007. *Benthic surveys of sandbanks in the Irish Sea*. Irish Wildlife Manuals, No. 29. National Parks and Wildlife Service, Department of Environment, Heritage and Local Government, Dublin, Ireland.
- Roddie, B., Kedwards, T., Ashby-Crane, R. & Crane, M., 1994. The toxicity to *Corophium volutator* (Pallas) of beach sand contaminated by a spillage of crude oil. *Chemosphere*, **29** (4), 719-727.
- Rosenberg, R., 1977. Benthic macrofaunal dynamics, production, and dispersion in an oxygen-deficient estuary of west Sweden. *Journal of Experimental Marine Biology and Ecology*, **26**, 107-33.
- Sanders, H.L., 1978. Florida oil spill impact on the Buzzards Bay benthic fauna: West Falmouth. *Journal of the Fisheries Board of*

Canada, **35**, 717-730.

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

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

Schottler, U. & Grieshaber, M., 1988. Adaptation of the polychaete worm *Scoloplos armiger* to hypoxic conditions. *Marine Biology*, **99** (2), 215-222.

SEEEC (Sea Empress Environmental Evaluation Committee), 1998. The environmental impact of the Sea Empress oil spill. *Final Report of the Sea Empress Environmental Evaluation Committee*, 135 pp., London: HMSO.

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

Shim, K.C., Koprivnikar, J. & Forbes, M.R., 2013. Variable effects of increased temperature on a trematode parasite and its intertidal hosts. *Journal of Experimental Marine Biology and Ecology*, **439**, 61-68.

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

Smyth, J.C., 1968. The fauna of a polluted site in the Firth of Forth. *Helgolander Wissenschaftliche Meeresuntersuchungen*, **17**, 216-233.

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

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

Sordino, P., Gambi, M.C. & Carrada, G.C., 1989. Spatio-temporal distribution of polychaetes in an Italian coastal lagoon (Lago Fusaro, Naples). *Cahiers de Biologie Marine*, **30**, 375-391.

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

Sundborg, Å., 1956. The River Klarälven: a study of fluvial processes. *Geografiska Annaler*, **38** (2), 125-237.

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

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

Thouzeau, G., Jean, F. & Del Amo, Y., 1996. Sedimenting phytoplankton as a major food source for suspension-feeding queen scallops (*Aequipecten opercularis* L.) off Roscoff (western English Channel) ? *Journal of Shellfish Research*, **15**, 504-505.

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

Ugolini, A., Ungherese, G., Somigli, S., Galanti, G., Baroni, D., Borghini, F., Cipriani, N., Nebbiai, M., Passaponti, M. & Focardi, S., 2008. The amphipod *Talitrus saltator* as a bioindicator of human trampling on sandy beaches. *Marine Environmental Research*, **65** (4), 349-357.

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

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

Van Colen, C., Montserrat, F., Vincx, M., Herman, P.M.J., Ysebaert, T. & Degraer, S., 2010. Long-term divergent tidal flat benthic community recovery following hypoxia-induced mortality. *Marine Pollution Bulletin* **60** (2), 178-186.

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

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

Vorobyova, L., Bondarenko, O. & Izaak, O., 2008. Meiobenthic polychaetes in the northwestern Black Sea. *Oceanological and Hydrobiological Studies*, **37** (1), 43-55.

Watkin, E.E., 1941. The yearly life cycle of the amphipod, *Corophium volutator*. *The Journal of Animal Ecology*, **10**, 77-93.

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

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

Wilding T. & Hughes D., 2010. A review and assessment of the effects of marine fish farm discharges on Biodiversity Action Plan



habitats. *Scottish Association for Marine Science, Scottish Aquaculture Research Forum (SARF)*.

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

Ysebaert, T., Meire, P., Maes, D. & Buijs, J., 1993. The benthic macrofauna along the estuarine gradient of the Schelde estuary. *Netherlands Journal of Aquatic Ecology*, **27** (2-4), 327-341.