Sublittoral sand in low or reduced salinity (lagoons)

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

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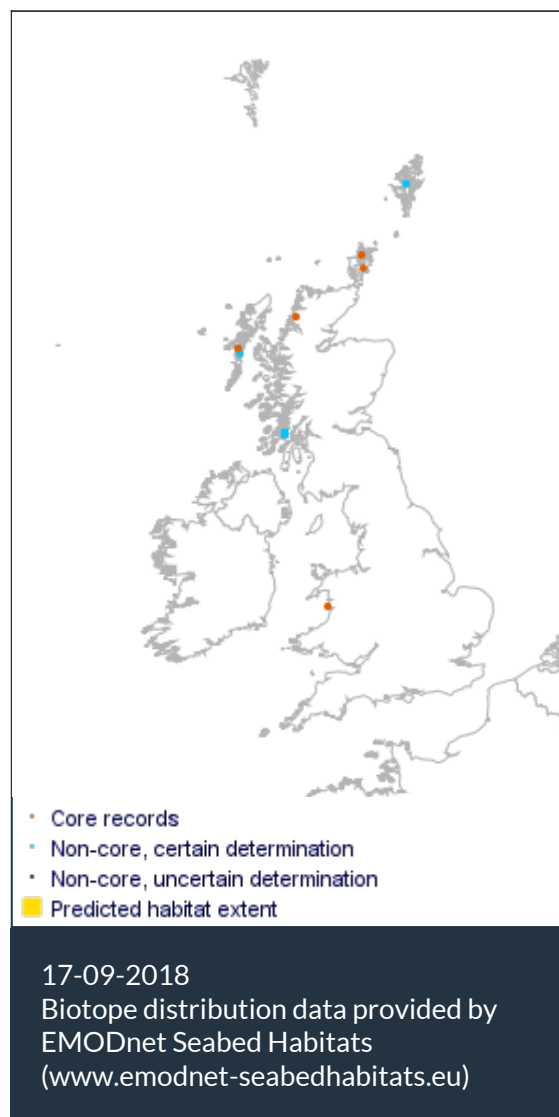
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- Core records
- Non-core, certain determination
- Non-core, uncertain determination
- Predicted habitat extent

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

Researched by   Dr Heidi Tillin & Dr Harvey Tyler-Walters   **Refereed by**   Admin

Summary

☰ UK and Ireland classification

EUNIS 2008	A5.21	Sublittoral sand in low or reduced salinity
JNCC 2015	SS.SSa.SSaLS	Sublittoral sand in low or reduced salinity (lagoons)
JNCC 2004	SS.SSa.SSaLS	Sublittoral sand in low or reduced salinity (lagoons)
1997 Biotope		

🏷 Description

Shallow sand and muddy sand in areas of low or reduced, although relatively stable salinity (may vary annually), with largely ephemeral faunal communities. The species are often similar to that found in SMuLS and are characterized by *Arenicola marina* with other species, including mysids, tubificoid and enchytraeid oligochaetes, *Corophium volutator*, *Hediste diversicolor*, *Pygospio elegans*, *Hydrobia ulvae* and *Cerastoderma glaucum*, which commonly occur in lagoons. Filamentous green algae such as *Chaetomorpha linum* may also be present. In some examples of this biotope the

polychaete *Fabricia stellaris* may be super-abundant and the isopod *Sphaeroma hookeri* common (JNCC, 2015).

↓ Depth range

-

🏛 Additional information

-

✔ Listed By

- none -

🔗 Further information sources

Search on:

G ɡ G JNCC

Sensitivity review

Sensitivity characteristics of the habitat and relevant characteristic species

The biotope description and characterizing species are taken from JNCC (2015). Shallow sand and muddy sand in areas of low or reduced, although relatively stable salinity (may vary annually), with largely ephemeral faunal communities. The species are often similar to that found in SMuLS and are characterized by *Arenicola marina* with other species, which commonly occur in lagoons. The sensitivity assessments focus on *Arenicola marina* and the characterizing oligochaetes and other associated species, including *Corophium volutator*, *Hediste diversicolor*, *Pygospio elegans*, *Hydrobia ulvae* and *Cerastoderma glaucum*.

Resilience and recovery rates of habitat

Saline lagoons are defined as areas of typically (but not exclusively) shallow, coastal saline water, wholly or partially separated from the sea by sandbanks, shingle or, less frequently, rocks or other hard substrata. They retain a proportion of their water at low tide and may develop as brackish, fully saline or hyper-saline water bodies (Brown *et al*., 1997). There are a number of factors which are thought to contribute to the rarity of these habitats in the UK. Many sedimentary lagoons do not appear in many of the areas suitable for them because of the macro-tidal regimes found within the north east Atlantic (Barnes, 1991). The high energy coastlines common around the UK mean that shingle lagoons can be both created from off-shore glacial deposits and removed, therefore they are often short lived on geological time scales, and are now less common than they were at the end of the last ice age (Bamber *et al*., 2001).

Recovery of impacted lagoon assemblages may occur through repair of damaged individuals, active migration or passive water transport od adults and juveniles and recolonization by pelagic larvae. A number of characterizing species, including *Arenicola marina*, *Cerastoderma glaucum*, and *Hediste diversicolor* produce pelagic larvae. Dispersal in *Corophium volutator* and *Tubificoides* may be more limited as these species brood young and produce egg cocoons rather than releasing pelagic larvae. *Pygospio elegans* and *Hydrobia ulvae* may produce either benthic or planktonic larvae/juveniles. Restricted water movements within lagoon system may prevent loss of larvae but could also reduce water transport of larvae into the biotope to replenish lost populations. Local hydrodynamics will strongly influence larval transport. Key life history factors and examples of recovery are outlined below for the key species. *Arenicola marina*, *Cerastoderma glaucum* and *Hediste diversicolor* are larger, longer-lived species and may structure the biotope through sediment disturbance and predation. Their populations may take longer to recover to typical age and biomass structures than those of the short-lived, more opportunistic *Hydrobia ulvae*, *Corophium volutator* and *Pygospio elegans*. The oligochaetes are intermediate between the two groups as they are longer-lived but their populations may be abundant and they are likely to recover rapidly (see below).

Arenicola marina has a high fecundity and spawns synchronously within a given area, although the spawning period varies between areas. Spawning usually coincides with spring tides and fair weather (high pressure, low rainfall and wind speed) (see *Arenicola marina* review). Eggs and early larvae develop within the female burrow. Post-larvae are capable of active migration by crawling, swimming in the water column and passive transport by currents (Farke & Berghuis, 1979) e.g. Günther (1992) suggested that post-larvae of *Arenicola marina* were transported distances in the range of 1 km. Juvenile settlement is density dependent and the juveniles avoid areas of high adult abundance and settle above the adults on the shore (Farke & Berghuis, 1979; Reise *et al*., 2001).

The juveniles migrate down the shore before or during the next winter, leaving the upper shore for the next generation. Reise *et al.* (2001) suggested that the largest and possibly oldest individuals were found seaward and in subtidal sands.Adults reach sexual maturity by their second year (Newell, 1948; Wilde & Berghuis, 1979) but may mature by the end of their first year in favourable conditions depending on temperature, body size, and hence food availability (Wilde & Berghuis, 1979). Beukema & de Vlas, (1979) suggested a lifespan, in the Dutch Wadden Sea, of at least 5-6 years, and cite a lifespan of at least 6 years in aquaria. They also suggested an average annual mortality or 22%, an annual recruitment of 20% and reported that the abundance of the population had been stable for the previous 10 years. However, Newell (1948) reported 40% mortality of adults after spawning in Whitstable.

McLusky *et al.* (1983) examined the effects of bait digging on blow lug populations in the Forth estuary. Dug and in-filled areas and unfilled basins left after digging re-populated 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. Beukema (1995) noted that the lugworm stock recovered slowly from mechanical dredging reaching its original level in at least three years. Reise *et al.* (2001) noted that a 50% reduction in the abundance of adult lugworm on sand flats in Sylt after the severe winter of 1995/96, was replaced by an enhanced recruitment of juveniles in spring, so that the effect of the severe winter on *Arenicola marina* population was small and brief. Beukema (1995) estimated that four to five years of mechanical dredging in the Balgzand region of the Wadden Sea, increased the mortality of the *Arenicola* population by ca 17% per year to a total of ca 40% per year and resulted in a long-term decline in the lugworm stock, until the dredge moved to a richer area. However, Beukema (1995) noted that the lugworm stock recovered slowly after mechanical dredging, reaching its original level after at least three years.

The recovery of *Arenicola marina* populations is generally regarded as rapid and occurs by recolonization by adults or colonization by juveniles from adjacent populations or the subtidal. However, Fowler (1999) pointed out that recovery may take longer on a small pocket, isolated, beach with limited possibility of recolonization from surrounding areas. Therefore, if adjacent populations are available recovery will be rapid. However, where the affected population is isolated or severely reduced, then recovery may be extended.

The cockle, *Cerastoderma glaucum* is a filter feeding bivalve which burrows shallowly in soft sediments and produces pelagic larvae. The species exists in two forms, a typical thin-shelled variety found in brackish lagoon habitats and a thicker shelled variety occurring in estuaries. It is widely distributed in north-west Europe. Limited evidence was found for recruitment in *Cerastoderma glaucum*. The congener *Cerastoderma edule* has been more extensively studied and general patterns are probably similar between the two species. A number of factors have been identified that affect larval supply and recruitment to adult populations of *Cerastoderma edule*. Survival during the first few months of life appears to be the decisive factor for recruitment success (Beukema & Dekker, 2005). High levels of juvenile recruitment have been observed where previous severe winters with heavy storm surges have reduced the population density of adults and reduced numbers of infaunal predators (Ducrotoy *et al.*, 1991).

The ability of postlarvae and larger juveniles and adults of *Hediste diversicolor* to swim, burrow and be carried by bedload transport can aid the rapid recolonization of disturbed sediments. (Shull, 1997). Davey & George (1986), found evidence that larvae of *Hediste diversicolor* were tidally dispersed within the Tamar estuary over a distance of 3 km, suggesting recolonization of disturbed sediments, or populations removed from bait digging is likely to occur rapidly, depending

upon larvae transport pathways. *Hediste diversicolor* are more likely to occur in the late transitional and the equilibrium communities that rely on more stable sediments that have recovered from disturbance (Newell et al., 1998).

The amphipod *Corophium volutator* lives for a maximum of one year (Hughes, 1988) and females can have 2-4 broods in a lifetime (Conradi & Depledge, 1999). Populations in southerly areas such as the Dovey Estuary, Wales or Starrs Point, Nova Scotia have two reproductive episodes per year. Those populations in colder, more northerly areas such as the Ythan Estuary, Scotland or 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 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.

Hydrobia ulvae may live from just over 1 year up to 2.5 years. Minimum egg hatching time has been recorded as five days. Individuals hatching from eggs laid in spring can breed in autumn, whereas those hatching in autumn overwinter before breeding in spring. Between 7 and 50 eggs are laid and both planktotrophic and lecithotrophic larvae have been recorded (Fretter & Graham, 1994, Barnes, 1988, Clay, 1960, Barnes, 1990, Fish & Fish, 1974, Anderson, 1971, Sola, 1996, Pilkington, 1971). *Hydrobia ulvae* have high powers of regeneration to the extent that head structures can be re-grown suggesting that individuals can recover from damage (Gorbushin *et al*., 2001). Recovery from superficial damage may be rapid.

The polychaete *Pygospio elegans* exhibit a number of reproductive strategies (a trait known as poecilogony). Larvae may develop directly allowing rapid population increase in suitable patches or they may have a planktonic stage (allowing colonization of new habitats). Experimental defaunation studies have shown an increase in *Pygospio elegans*, higher than background abundances within 2 months, reaching maximum abundance within 100 days (Van Colen *et al*. 2008). Following a period of anoxia in the Bay of Somme (north France) that removed cockles, *Pygospio elegans* increased rapidly but then decreased as cockle abundance recovered and sediments were disturbed by cockle movement (Desprez *et al*., 1992). Recovery will depend on the lack of stronger competitors and the supply of larvae and hence the season of disturbance will moderate recovery time. In general recovery is predicted to occur within 6 months. However, where conditions are stable these species are likely to be replaced by competitive dominants, particularly bivalves. Like other polychaetes and molluscs *Pygospio elegans* may suffer from predation by fish and birds on exposed parts of the body and can rapidly repair this (repair takes between 9-12 days (Lindsay *et al*., 2007).

Tubificid populations tend to be large and to be constant throughout the year, although some studies have noticed seasonal variations (Giere & Pfannkuche, 1982). Many species,

including *Tubificoides benedii* and *Baltidrilus costata* have a two-year reproductive cycle and only part of the population reproduces each season (Giere & Pfannkuche, 1982). Populations of *Tubificoides benedii* in the Fourth estuary have not demonstrated clear seasonality in recruitment (Bagheri & McLusky, 1982), although mature *Tubificoides benedii* (as *Peloscolex benedeni*) in the Thames Estuary were reported to occur in December with a maximum in late February (Hunter & Arthur, 1978), breeding worms increased from April and maximum cocoon deposition was observed in July (Hunter & Arthur, 1978). It is dominant, often reaching huge population densities in coastal areas that are enriched in organic matter and is often described as an 'opportunist' species adapted to rapid environmental fluctuations and stress (Giere, 2006; Bagheri & McLusky, 1982). However, unlike other opportunist species it has a long lifespan (a few years, Giere, 2006), a prolonged reproductive period from reaching maturity to maximum cocoon deposition and exhibits internal fertilisation with brooding rather than pelagic dispersal. These factors mean that recolonization is slower than for some opportunistic species which may be present in similar habitats.

Bolam and Whomersley (2003) observed faunal recolonization of fine sediments placed on saltmarsh as a beneficial use and disposal of fine-grained dredged sediments. They found that tubificid oligochaetes began colonising sediments from the first week following a beneficial use scheme involving the placement of fine-grained dredged material on a salt marsh in southeast England. The abundance of *Tubificoides benedii* recovered slowly in the recharge stations and required 18 months to match reference sites and those in the recharge stations prior to placement of sediments. The results indicate that some post-juvenile immigration is possible and that an in-situ recovery of abundance is likely to require more than 1 year. Rapid recolonization has also been observed in the tubificid oligochaete *Baltidrilus costata (Tubifex costatus)* appeared in upper sediment layer s in experimentally defaunated patches (4m^2) after 3 weeks (Gamenick *et al*., 1996).

Resilience assessment. Overall, the recovery *potential* of *Arenicola marina* and the associated, polychaetes and oligochaetes and other species is probably rapid. It must be considered that the location of this biotope within lagoon habitats will limit the ability of this biotope to be quickly recolonized. The low flushing times and high water retention rates within saline lagoons means that larva and propagule supply will also be slow. *Arenicola marina* and populations of species such as oligochaetes which produce egg cocoons and *Corophium volutator* which brood young may be self-recruiting and recovery from some mortality may be rapid. However, should a population be severely reduced it may take some time for recolonization to occur from other populations. Therefore, where resistance is 'Medium' (some mortality) a resilience of **High** is recorded but where resistance is lower ('Low' to 'None'; significant mortality) a resilience of Medium (2-10 years) is recorded, to represent the isolated waters in which this biotope is found.

Lagoons are rare within the UK and, on geological timescales, are short-lived coastal features. Spencer & Brooks (2012) state that over decadal timescales, the seaward barriers that enclose saline lagoons naturally migrate progressively landwards. This is part of the natural succession of the habitat. However, these seaward barriers will move even faster if local geomorphological conditions change. A change in geomorphological conditions could both speed up the movement of the lagoon landwards, and change the supply of sediment to the barrier. Ultimately this could lead to the total removal of the barrier and the loss of the lagoon. The effects of an artificial structure built behind a lagoon must also be taken into consideration. It is possible that such a structure could potentially inhibit the natural migration of the lagoon land wards, and thus reduce the lifespan of the lagoon. Where a pressure is likely to remove the lagoon habitat recovery may be protracted or may not occur.

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)	Medium	High	Low
	Q: High A: High C: Medium	Q: High A: High C: High	Q: High A: High C: Medium

Lagoon environments are typically shallow (Barnes, 1994) and will, therefore, heat and cool more rapidly than deeper habitats or those that are more connected to the sea and tidally flushed. Species that characterize this habitat are therefore likely to be able to tolerate higher and lower temperatures than open coast species from the shallow subtidal. Populations may be acclimated to the temperatures or able to recover rapidly where populations are impacted.

Arenicola marina is recorded from shores of western Europe, Norway, Spitzbergen, north Siberia, and Iceland. In the western Atlantic, it has been recorded from Greenland, along the northern coast from the Bay of Fundy to Long Island. Its southern limit is about 40°N (see *Arenicola marina* review), although OBIS (2016) includes a few records from the Atlantic coast of Africa and the Mediterranean. Sommer *et al.* (1997) examined sub-lethal effects of temperature in *Arenicola marina* and suggested a critical upper and lower temperature of 20°C and 5°C respectively in North Sea specimens. Above or below these critical temperatures, specimens resort to anaerobic respiration. Sommer *et al.* (1997) noted that specimens could not acclimate to a 4°C increase above the critical temperature. De Wilde & Berghuis (1979) reported 20% mortality of juveniles reared at 5°C, negligible mortality at 10 and 15°C but 50% mortality at 20°C and 90% at 25°C.

Schroeer *et al.* (2009) identified a shift in the thermal tolerance of *Arenicola marina*, with an optimum moving towards higher temperatures with decreasing latitudes, suggesting the species may adapt to long-term shifts such as 2°C but over time. Therefore, *Arenicola marina* in UK and Irish populations will occupy an optimum temperature range in relation to UK and Irish latitudes. An upper limit above 20°C may occur in more southerly populations. In studies in Whitley Bay, Tyne and Wear, UK, *Arenicola marina* was most active in spring and summer months, with a mean rate of cast production fastest in spring and particularly slow in autumn and winter, suggesting feeding rate is greatest in higher temperatures (Retraubun *et al.*, 1996). Retraubun *et al.* (1996) also showed that cast production by specimens in lab experiments increased with temperature, peaking at 20°C before declining. Rates of cast production at 30°C were still higher than at 10°C, suggesting UK populations may have greater tolerance to higher temperatures than populations studied in more northerly latitudes. Temperature changes may affect maturation, spawning time and synchronisation of spawning and reproduction in the long-term (Bentley & Pacey, 1992; Watson *et al.*, 2000). Spawning can be inhibited in gravid adults maintained above 15°C (Watson *et al.*, 2000). However, spawning success would remain dependent upon spring and autumn

temperatures remaining below 15°C. Additionally, an impact from temperature change at the substratum surface may be mitigated as *Arenicola marina* is protected from direct effects by their position in the sediment.

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).

Cerastoderma glaucum occurs from Norway and the Baltic to the Mediterranean and Black Seas. *Cerastoderma glaucum* can tolerate a wide range of temperatures. Populations have been observed to survive in lagoons in conditions to more than 30°C.

The geographic range of *Hediste diversicolor* suggests that it is resistant of a range of temperatures and a temperature increase at benchmark levels is unlikely to have an adverse effect on UK populations. *Hediste diversicolor* can tolerate temperatures from below zero under Baltic ice to high summer temperatures (>20°C) in Black Sea lagoons (Smith, 1977). *Hediste diversicolor* were not strongly affected by heatwaves in an estuary in north western Portugal, where temperatures may reach 40°C in intertidal pools (higher temperatures than experienced around UK and Irish coasts) (Dolbeth *et al.* 2011). Temperature change may adversely affect reproduction. Bartels-Hardege and Zeeck (1990) demonstrated that an increase from 12°C and maintenance of water temperature at 16°C induced reproduction in *Hediste diversicolor* specimens outside the normal period of spawning, and without a drop in temperature to simulate winter conditions the spawning period was prolonged and release of gametes was not synchronized. Poor synchronization of spawning could result in reduced recruitment, as gametes are wasted and adults die shortly after gamete release.

Deeper burrowing oligochaetes are protected from fluctuations in temperature by the overlaying sediments which dampen changes if poorly drained (Giere & Pfannkuche, 1982). Bamber & Spencer (1984) observed that *Tubificoides* were dominant species in an area affected by thermal discharge in the River Medway estuary. Sediments were exposed to the passage of a temperature front of approximately 10°C between heated effluent and estuarine waters during the tidal cycles. *Pygospio elegans* were summer visitors to the areas affected by heated effluent.
Pygospio elegans also shows a relationship between timing of reproduction and temperature. Gibson and Harvey (2000) in a study on asexual reproduction of *Pygospio elegans* in Nova Scotia, Canada found temperature did not influence reproduction strategy (planktotrophy, lecithotrophy or asexual reproduction) but cite Anger (1984) that environmental conditions, including temperature influence timing of reproduction.

Sensitivity assessment. Based on species distributions and evidence for temperature tolerance, the characterizing species are likely to be tolerant of a chronic increase in temperature at the pressure benchmark. *Arenicola marina* and the associated species are unlikely to be removed entirely but may be impacted by a short-term acute change in temperature of 5°C. Hence, a resistance of **Medium** is suggested to represent a loss of some of the population. Resilience is probably **High** and sensitivity is assessed as **Low**.

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

Lagoon environments are typically shallow (Barnes, 1994) and will, therefore, heat and cool more

rapidly than deepen habitats or those that are more connected to the sea and tidally flushed. Species that characterize this habitat are therefore likely to be able to tolerate higher and lower temperatures than open coast species from the shallow subtidal. Populations may be acclimated to the temperatures or able to recover rapidly where populations are impacted.

Arenicola marina is recorded from shores of western Europe, Norway, Spitzbergen, north Siberia, and Iceland. In the western Atlantic, it has been recorded from Greenland, along the northern coast from the Bay of Fundy to Long Island. Its southern limit is about 40°N (see *Arenicola marina* review), although OBIS (2016) includes a few records from the Atlantic coast of Africa and the Mediterranean.

Arenicola marina displays a greater tolerance to decreases in temperature than to increases, although optimum temperatures are reported to be between 5°C and 20°C. Reise *et al*. (2001) stated that *Arenicola marina* was known to be a winter hardy species and that its abundance and biomass were stable even after severe winters. Sommer *et al*. (1997) report populations in the White Sea (sub-polar) acclimatised to -2°C in winter. Populations in the North Sea (boreal) were less tolerant of temperatures below 5°C, although in laboratory experiments on individual lugworms from North Sea populations worms survived a temperature drop from 6 or 12°C to -1.7°C for more than a week (Sommer & Portner, 1999).

Temperature change may affect maturation, spawning time and synchronisation of spawning and reproduction in the long-term (Bentley & Pacey, 1992; Watson *et al*., 2000). Spawning success is dependent upon spring and autumn temperatures, the seasons when spawning occurs in relation to spring and neap tides, remaining below 13-15°C. De Wilde & Berghuis (1979) reported 20% mortality of juveniles reared at 5 °C, negligible mortality at 10 °C and 15 °C but 50% at 20°C and 90% mortality at 25°C.

Evidence from the Sylt in the North Sea suggests that the effects of severe winters on *Arenicola marina* populations are small and brief (Reise, et al., 2001) The severe winter of 1995/1996 disrupted the usual juvenile settlement cycle in the sand flats of the Sylt, North Sea (Reise et al., 2001). In the severe winter, the adult population of *Arenicola marina* migrated down the shore, to deeper, waters to avoid low temperatures and 66 days of ice on the intertidal sand flats. Although, the adult population was halved, and no dead lugworms were observed on the surface or in the sediment. The post-larvae hibernate in the deep water channels (subtidal) in shell gravel and mussel beds. In summer the juveniles were not restricted to the upper shore but settled over a wider area of the flats, in the space left by the adult population. Reise *et al*. (2001) concluded that the enhanced recruitment demonstrated that the post-larvae did not suffer increased mortality during the winter, probably as their subtidal hibernation sites did not experience ice cover. Similarly, *Arenicola marina* was listed as 'apparently unaffected' by the severe 1962/63 winter in the UK (Crisp, 1964).

The geographic range of *Hediste diversicolor* suggests that it is resistant of a range of temperatures and a temperature increase at benchmark levels is unlikely to have an adverse effect on UK populations. *Hediste diversicolor* can tolerate temperatures from below zero under Baltic ice to high summer temperatures (>20°C) in Black Sea lagoons (Smith, 1977).

Cerastoderma glaucum occurs from Norway and the Baltic to the Mediterranean and Black Seas. *Cerastoderma glaucum* can also tolerate a wide range of temperatures. Populations have been observed to survive in lagoons in conditions from almost freezing to more than 30 °C. However, a high mortality of cockles was observed in South East England after the exceptionally cold winter of

1962/3. Boyden (1972) observed that *Cerastoderma glaucum* has a LD50 of 62.3 hours at 20 degrees C in air. *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).

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

Crisp (1964) reported that species of amphipod and isopods seemed to be unharmed by the severe winter of 1962-1963. This may be due to burial in sediments buffering temperature or seasonal migration to deeper waters to avoid freezing. *Corophium volutator* is subject to temperatures of 1°C in the winter to 17°C in the summer (Wilson & Parker, 1996). The population may reduce activity and delay reproduction if the temperature drops below 7°C. Sudden pulses of very cold water can disrupt the circa-tidal rhythms of *Corophium volutator* by resetting the onset of swimming behaviour. For example, a 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).

Sensitivity assessment. *Arenicola marina* and the populations of other species present in the biotope are distributed to the north of the British Isles, exhibit regional acclimation to temperature, are known to be winter hardy, and can migrate to deeper waters or within sediments to avoid change in temperature and even ice. Therefore, the biotope is probably resistant of a short to long-term decrease in temperature at the benchmark level and a resistance of **High** is suggested. Hence, resilience is **High** and the biotope is assessed as **Not sensitive** at the benchmark level.

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

This biotope occurs in reduced (18-35 pp) and low (<18 ppt) salinity (JNCC, 2015). Biotopes that occur in low salinity are considered tolerant of an increase to reduced salinity as this falls within the habitat range. A change at the pressure benchmark is assessed as a change to full salinity. Species in lagoons may be able to tolerate some fluctuations in salinity as evaporation on hot days may increase salinity and tidal flushing may lead to some changes.

Cerastoderma glaucum can survive in lagoons where salinity varies from 7 psu to 84 psu, so it has a high tolerance of different salinities, however a change to full salinity may lead to replacement by *Cerastoderma edule*.

Arenicola marina loses weight when exposed to hyperosmotic shock (47 psu for 24 hrs) but are able to regulate and gain weight within 7-10 days (Zebe & Schiedek, 1996).

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

Oligochaete dominated biotopes are recorded from a range of salinity regimes from full (LS.LSa.MoSa.Ol; LS.LSa.MoSa.Ol.FS), variable (SS.SMu.SMuVS.CapTubi) to low (SS.SMu.SMuVS.LhofTtub) habitats (JNCC,2015). The species characterizing these biotopes are likely to vary. Giere & Pfannkuche (1982) identified how species change over a hypothetical salinity gradient with marine stenohaline species present at full salinities replaced by more euryhaline oligochaete species including *Tubificoides benedii* and *Tubificoides pseudogaster*, *Paranais litoralis* and *Baltidrilus costata* (formerly *Heterochaeta costata*). Studies in the Rhine delta have found that *Tubificoides benedii*, is more tolerant of a range of salinities than *Baltidrilus costata* (as *Heterochaeta costata*) which preferred shallow water brackish stations (Verdonschot *et al*. 1982). However, numerous studies suggest that *Baltidrilus costata* tolerates a wide range of salinities from 1‰ to 28‰ (Giere & Pfannkuche, 1982 and references therein), suggesting that while tolerant of some changes, an increase to full salinity may lead to reductions in abundance of this species.

Sensitivity assessment. The characterizing species are able to survive in both fully marine and brackish water and may be unaffected by an increase in salinity (although some acclimation may be required). As salinity is a key structuring factor it is likely that a long-term change in salinity would alter the biological assemblage allowing species that are intolerant of low and reduced salinity to colonize. Biotope resistance is therefore assessed as **Low** and resilience as **Medium**, so that sensitivity to this pressure is assessed as **Medium**.

Salinity decrease (local)	None	Medium	Medium
	Q: Low A: NR C: NR	Q: Low A: NR C: NR	Q: Low A: Low C: Low

This biotope occurs in reduced (18-35 pp) and low (<18 ppt) salinity (JNCC, 2015). Biotopes that occur in reduced salinity are considered tolerant of a change to low as this fall within the habitat range. A change at the pressure benchmark is assessed as a change to freshwater. Few of the characterizing species are likey to tolerate a long-term salinity reduction at the pressure benchmark. Biotope resistance is therefore assessed as **None** and resilience as **Medium** (following habitat recovery). Biotope sensitivity is therefore assessed as **Medium.**

Water flow (tidal current) changes (local)	None	High	Medium
	Q: Low A: NR C: NR	Q: Low A: NR C: NR	Q: Low A: Low C: Low

This biotope is found in very weak tidal conditions where water flow is negligible (JNCC, 2015). An increase in water flow at the benchmark is unlikely to have a significant negative effect on the biological composition of the biotope as many of the characterizing species are found in biotopes that are exposed to stronger tidal streams. However, it must be assumed that if there is an increase in the water flow within the biotope there will also be an increase in the water flow around the lagoon. An increase or decrease in the water flow at the pressure benchmark may result in increased erosion of sands or deposition of fine silts resulting in changes in sediment composition. An increase in water flow above the level of the benchmark this could consequently

lead to the loss/collapse of a lagoon wall and completely change the physical environment of the biotope, and lead to the loss of the lagoon biotope altogether. Decreased water flow may also have implications for oxygenation and larval transport as water exchange and flushing are reduced. General predictions are limited as the change will depend on site-specific factors such as the structure of the lagoon, the degree of water exchange and sediment supply.

Sensitivity assessment. The biotope occurs in weak to very weak flow so that any further reduction is not relevant. An increase in water flow could modify the sediment. A significant increase may result in a change in the sediment from fine muds to sand muds and the fine particulates are removed. The experimental evidence suggests that a change in the flow of 0.11 m/s to 0.25 m/s was enough to alter the sediment and the appearance of the biotope within 65 days. Therefore, a reduction in flow of 0.1-0.2 m/s may alter the sediment and the appearance of the biotope resulting in a change to the mud lagoon SS.SMu.SMuLS. Therefore, a resistance of **None** is suggested. Resilience is probably **High** as, once the low energy conditions return, it may only take a couple of years for the mud to deposit in these otherwise sheltered and isolated habitats. Hence, sensitivity is assessed as **Medium**.

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

Changes in emergence is only relevant to intertidal and sublittoral fringe biotopes.

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

This biotope is found in extremely sheltered conditions. An increase or decrease in wave exposure at the benchmark is unlikely to have a significant negative effect on the biological composition of the biotope. However, it might be assumed that if there is an increase in the wave height within the biotope there will also be an increase in the water flow around the lagoon. This increase could have a detrimental effect on the structure of the lagoon through erosion. This could consequently lead to the loss collapse of a lagoon wall and completely change the physical environment of the biotope, and lead to the loss of the lagoon biotope altogether.

Sensitivity assessment. Wave exposure can have a significant structuring effect on biotopes through direct effects on species and indirectly through changes in sediment composition and transport and physical disturbance. As lagoons are sheltered from waves a change at the pressure benchmark is very small. Resistance and resilience are assessed as 'High' and the biotope is assessed as 'Not Sensitive'.

⚗ Chemical Pressures

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

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

Contamination at levels exceeding the pressure benchmark may have negative effects. A 2-year microcosm experiment was undertaken to investigate the impact of copper on the benthic fauna of the lower Tyne Estuary (UK) by Hall & Frid (1995). During a 1-year simulated contamination period, 1 mg l⁻1copper was supplied at 2-weekly 30% water changes, at the end of which the sediment concentrations of copper in contaminated microcosms reached 411 µg g⁻1. Toxicity effects reduced populations of the four dominant taxa including *Tubificoides* spp.). When copper dosage was ceased and clean water supplied, sediment copper concentrations fell by 50% in less than 4 days, but faunal recovery took up to 1 year, with the pattern varying between taxa. Since the copper leach rate was so rapid it is concluded that after remediation, contaminated sediments show rapid improvements in chemical concentrations, but faunal recovery may be delayed taking up to a year.

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

Sediment may act as a sink for heavy metals contamination so that deposit feeding species may be particularly vulnerable to heavy metal contamination through ingestion of particulates. At high concentrations of Cu, Cd or Zn the blow lug left the sediment (Bat & Raffaelli, 1998). The following toxicities have been reported in *Arenicola marina*:

- no mortality after 10 days at 7 µg Cu/g sediment, 23 µg Zn/g and 9 µg Cd/g;
- median lethal concentrations (LC_{50}) of 20 µg Cu/g, 50 µg Zn/g, and 25 µg Cd/g (Bat & Raffaelli, 1998).

Bryan (1984) suggested that polychaetes are fairly resistant to heavy metals, based on the species studied. Short-term toxicity in polychaetes was highest to Hg, Cu and Ag, declined with Al, Cr, Zn and Pb whereas Cd, Ni, Co and Se the least toxic.

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

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

Sheltered embayments and lagoons, where this biotope is found, are particularly vulnerable to oil pollution, which may settle onto the sediment and persist for years (Cole *et al.*, 1999). Subsequent digestion or degradation of the oil by microbes may result in nutrient enrichment and eutrophication (see nutrients below). Although protected from direct smothering by oil by its depth, the biotope is relatively shallow and would be exposed to the water soluble fraction of oil, water soluble PAHs, and oil adsorbed onto particulates.

Suchanek (1993) reviewed the effects of oil spills on marine invertebrates and concluded that, in general, on soft sediment habitats, infaunal polychaetes, bivalves and amphipods were particularly affected. Crude oil and oil: dispersant mixtures were shown to cause mortalities in *Arenicola marina* (see review). Prouse & Gordon (1976) found that blow lug was driven out of the sediment by waterborne fuel oil concentrations of >1 mg/l or sediment concentration of >100 µg/g. Crude oil and refined oils were shown to have little effect on fertilization in sea urchin eggs but in the presence of dispersants, fertilization was poor and embryonic development was impaired (Johnston, 1984). Sea urchin eggs showed developmental abnormalities when exposed to 10-30mg/l of hydrocarbons and crude oil : Corexit dispersant mixtures have been shown to cause functional loss of tube feet and spines in sea urchins (Suchanek, 1993). Although no direct information on synaptid holothurians was found, it seems likely that adults and especially larvae are intolerant of hydrocarbon contamination.

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.

Tubificoides benedii appears to be more tolerant and was found in UK waters near oil refineries as the sole surviving member of the macrofauna. Populations were however apparently reduced and the worms were absent from areas of oil discharge and other studies indicate sensitivity to oiling (Giere & Pfannkuche, 1982, references therein).

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

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

The xenobiotic ivermectin was used to control parasitic infestations in livestock including sea lice in fish farms, degrades slowly in marine sediments (half-life >100 days). Ivermectin was found to produce a 10 day LC50 of 18µg ivermectin /kg of wet sediment in *Arenicola marina*. Sub-lethal effects were apparent between 5 - 105 µg/kg. Cole *et al*. (1999) suggested that this indicated a high intolerance. *Arenicola marina* has shown negative responses to chemical contaminants, including damaged gills following exposure to detergents (Conti, 1987), and inhibited the action of esterases following suspected exposure to point source pesticide pollution in sediments from the Ribble estuary, UK (Hannam *et al*., 2008).

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

ethanol if exposed for 24 hours or more but can withstand higher concentrations in short pulses. Such short pulses, however, have the effect of rephasing the diel rhythm and will delay the timing of swimming activity for the duration of the ethanol pulse (Harris & Morgan, 1984b).

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

Reports on littoral sediment benthic communities at Sandside Bay, adjacent to Dounray nuclear facility, Scotland, (where radioactive particles have been detected and removed) reported *Arenicola marina* were abundant (Dounray Particle Advisory Group, 2008). Kennedy *et al.* (1988) reported levels of ^{137}Cs in *Arenicola spp.* of 220-440 Bq/kg from the Solway Firth. However, no information on the effects of radionuclide contamination was found.

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	High Q: High A: High C: High	High Q: High A: High C: High	Not sensitive Q: High A: High C: High

Species that are likely to be sensitive to de-oxygenation at the pressure benchmark are juvenile *Cerastoderma glaucum*, *Pygospio elegans* and *Corophium volutator*. *Pygospio elegans* is highly sensitive to hypoxia (Gogina *et al.*, 2010). Exposure to dissolved oxygen concentration of less than or equal to 2 mg/l for 1 week is likely to limit *Pygospio elegans* abundance. Juvenile *Cerastoderma glaucum* are also very intolerant of low oxygen concentrations. However adults can survive for 84 hours in oxygen free water (Boyden, 1972). In a series of experiments Gamenick *et al.*, (1996) suggested that *Corophium volutator* is highly sensitive to hypoxia and suffers 50% mortality after just 4 hours in hypoxic conditions, or in 2 hours if there is rapid build-up of sulphide (Gamenick *et al.*, 1996). These results are largely in concordance with other work by Gamble (1970) who found that survival rates were temperature dependent with individuals surviving longer at lower temperatures. The level of oxygen was not assessed by Gamenick *et al.*, 1996) and the description of the experimental set-up suggests that anoxic test conditions were used rather than hypoxic. Gamble (1970) found that at 5°C most individuals were inactive after 30 minutes exposure to anaerobic seawater and that mortality occurred later, the inactivity may have allowed the species to survive longer (Gamble, 1970).

Other species within the biotope are more tolerant of hypoxia. *Tubificoides benedii* has a high capacity to tolerate anoxic conditions, its extreme oxygen tolerance is based on an unusually low respiration rate (Giere *et al.*, 1999). Respiration rates of *Tubificoides benedii* measured at various oxygen concentrations showed that aerobic respiration is maintained even at very low oxygen concentrations (Giere *et al.*, 1999). Birtwell & Arthur (1980) showed that *Tubificoides benedii* could tolerate anoxia in the Thames Estuary (LT_{50} = 58.8 hours at 20°C, 26.6 hours at 25°C and 17.8 hours at 30°C in experiments with worms acclimated to 20°C.)

Hediste diversicolor is resistant to moderate hypoxia (Diaz & Rosenberg, 1995). Vismann (1990) demonstrated a mortality of only 15% during a 22 day exposure of *Hediste diversicolor* at 10% oxygen (ca. 2.8 mg O_2 per litre). *Hediste diversicolor* is active at the sediment/water interface where

hydrogen sulphide concentrations increase during periods of hypoxia. Vismann (1990) also demonstrated that the high tolerance of *Hediste diversicolor* to hypoxia in the presence of sulphide is enabled by elevated sulphide oxidation activity in the blood. *Hediste diversicolor* may also exhibit a behavioural response to hypoxia by leaving the sediment (Vismann, 1990) in the presence of sulphide. After 10 days of hypoxia (10% oxygen saturation) with sulphide (172-187 µmM) only 35% of *Hediste diversicolor* had left the sediment compared to 100% of *Nereis virens*. Laboratory experiments in the absence of sediments, found that *Hediste diversicolor* could survive hypoxia for more than 5 days and that it had a higher tolerance to hypoxia than *Nereis virens*, *Nereis succinea* and *Nereis pelagica* (Theede, 1973; Dries & Theede, 1974; Theede *et al.*, 1973). Juvenile *Hediste diversicolor* survived hypoxic conditions for 4 days in laboratory conditions and combined hypoxia and increased sulphide (1 mmo1 1-') for 3 days (Gamenick *et al.*, 1996). Post larvae *Hediste diversicolor* were the only life stage to show les tolerance to hypoxia, surviving for only 14 hr (Gamenick *et al.*, 1996).

Arenicola marina is subject to reduced oxygen concentrations regularly at low tide and is capable of anaerobic respiration. The transition from aerobic to anaerobic metabolism takes several hours and is complete within 6-8 hrs, although this is likely to be the longest period of exposure at low tide. Fully aerobic metabolism is restored within 60 min once oxygen returns (Zeber & Schiedek, 1996). This species was able to survive anoxia for 90 hrs in the presence of 10 mmol/l sulphide in laboratory tests (Zeber & Schiedek, 1996). Hydrogen sulphide (H_2S) produced by chemoautotrophs within the surrounding anoxic sediment and may, therefore, be present in *Arenicola marina* burrows. Although the population density of *Arenicola marina* decreases with increasing H_2S, *Arenicola marina* is able to detoxify H_2S in the presence of oxygen and maintain a low internal concentration of H_2S. At high concentrations of H_2S in the lab (0.5, 0.76 and 1.26 mmol/l) the lugworm resorts to anaerobic metabolism (Zeber & Schiedek, 1996). At 16°C *Arenicola marina* survived 72 hrs of anoxia but only 36 hrs at 20°C. Tolerance of anoxia was also seasonal, and in winter anoxia tolerance was reduced at temperatures above 7°C. Juveniles have a lower tolerance of anoxia but are capable of anaerobic metabolism (Zebe & Schiedek, 1996). However, *Arenicola marina* has been found to be unaffected by short periods of anoxia and to survive for 9 days without oxygen (Borden, 1931 and Hecht, 1932 cited in Dales, 1958; Hayward, 1994). Diaz & Rosenberg (1995) listed *Arenicola marina* as a species resistant of severe hypoxia.

Sensitivity assessment. Burrowing species such as *Arenicola marina*, *Hediste diversicolor* and the oligochaetes burrow into anoxic sediment and may be tolerant of hypoxia. Species that occur at the surface and that are frequently found in well-oxygenated sandy sediments such as *Pygospio* elegans and *Corohpium volutator* are likely to be more sensitive. At the pressure benchmark biotope resistance is assessed as **Low** based on the loss of some characterizing species, biotope resilience is assessed as **High** and sensitivity is **Low.**

Nutrient enrichment	Not relevant (NR)	Not relevant (NR)	Not sensitive
	Q: NR A: NR C: NR	Q: NR A: NR C: NR	Q: NR A: NR C: NR

The abundance and biomass of *Arenicola marina* increase with increased organic content in their favoured sediment (Longbottom, 1970; Hayward, 1994). Therefore, moderate nutrient enrichment may be beneficial.

Indirect effects may include algal blooms and the growth of algal mats (e.g. of *Ulva* sp.) on the surface of the intertidal flats. Algal mats smother the sediment, and create anoxic conditions in the sediment underneath, changes in the microphytobenthos, and with increasing enrichment, a

reduction in species richness, the sediment becoming dominated by pollution tolerant polychaetes, e.g. *Manayunkia aestuarina*. In extreme cases, the sediment may become anoxic and defaunated (Elliot *et al.*, 1998). Algal blooms have been implicated in mass mortalities of lugworms, e.g. in South Wales where up to 99% mortality was reported (Boalch, 1979; Olive & Cadman, 1990; Holt *et al.* 1995). Feeding lugworms were present and exploitable by bait diggers within 1 month, suggesting rapid recovery, probably by migration from surrounding areas or juvenile nurseries.

Nevertheless, this biotope is considered to be **Not sensitive** at the pressure benchmark that assumes compliance with good status as defined by the WFD.

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

This biotope is characterized by species that are exclusively deposit feeders or that can switch to deposit feeding, including *Arenicola marina*, *Hediste diversicolor*, *Pygospio elegans,* oligochaetes and *Corophium volutator*. An input of organic matter may provide a food subsidy for these species.

The abundance and biomass of *Arenicola marina* increase with increased organic content in their favoured sediment (Longbottom, 1970; Hayward, 1994). Moderate enrichment increases food supplies, enhancing productivity and abundance. Gray *et al.* (2002) concluded that organic deposits between 50 to 300 gC m^{-2} yr^{-1}, are efficiently processed by benthic species. Substantial increases > 500 g C m^{-2} yr^{-1} would likely to have negative effects, limiting the distribution of organisms and degrade the habitat, leading to eutrophication, algal blooms, and changes in community structure to a community dominated by opportunist species (e.g. capitellids) with increased abundance but reduced species richness, and eventually to abiotic anoxic sediments (Pearson & Rosenberg, 1978; Gray, 1981; Snelgrove *et al.*,1995; Cromey *et al.*, 1998).

Borja *et al.* (2000) and Gittenberger & loon (2011) placed *Arenicola marina* into the AMBI pollution group III, defined as 'Species tolerant to excess organic matter enrichment. These species may occur under normal conditions, but their populations are stimulated by organic enrichment (slight unbalance situations)'.

Tubificoides benedii and *Baltidrilus costatus* are both very tolerant of high levels of organic enrichment and often dominate sediments where sewage has been discharged or other forms of organic enrichment have occurred (Pearson & Rosenberg, 1978; Gray, 1971; McLusky *et al.*, 1980). Their tolerance for organic enrichment is attributed to their adaptation to live in and feed on enriched organic deposits (Pearson & Rosenberg, 1978) and their high population densities in such areas is enhanced by the lack of predation and competition. *Tubificoides benedii* are abundant in mussel beds (mussel relaying may be the source of smothering) which has been attributed to their tolerance of organically rich deoxygenated sediment (Commito & Boncavage, 1989). *Tubificoides benedii* has also been found in elevated abundances in areas of organic enrichment around fish farms (Haskoning, 2006).

Sensitivity assessment. The biotope is probably rich in organic matter as it occurs in sheltered, isolated areas. An input of organic matter at the pressure benchmark is likely to provide food for deposit feeders within the biotope. Therefore, a resistance of **High** is suggested at the benchmark level. Hence, resilience is **High** and the biotope is assessed as **Not sensitive** at the benchmark level.

⚓ Physical Pressures

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

All marine habitats and benthic species are considered to have a resistance of 'None' to this pressure and to be unable to recover from a permanent loss of habitat (resilience is 'Very Low'). Sensitivity 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.

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

The biotope is characterized by the sedimentary lagoon habitat (JNCC, 2015), so a change to an artificial or rock substratum would alter the character of the biotope leading to reclassification and the loss of the sedimentary community including the characterizing bivalves, polychaetes and echinoderms that live buried within the sediment.

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

	Resistance	Resilience	Sensitivity
Physical change (to another sediment type)	None Q: High A: High C: High	Very Low Q: High A: High C: High	High Q: High A: High C: High

A change in sediment type by one Folk class (using the Long 2006 simplification) would change the sediment to mud or mixed sediments. Although many of the characterizing species are likely to persist, the biotope would probably be re-classified to the sublittoral mud lagoon type SS.SMu.SMuLS or the mixed sediment SS.SMx.SMxLS. A change to coarse sediments would significantly alter species composition through loss of *Arenicola marina* and amphipods and oligochaetes and other associated species. Therefore, a resistance of **None** is recorded, resilience is **Very low** (the pressure is a permanent change) and sensitivity is assessed as **High**.

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

Sedimentary communities are likely to be highly intolerant of substratum removal, which will lead to partial or complete defaunation, exposing underlying sediment which may be anoxic and/or of a different character or bedrock and lead to changes in the topography of the area (Dernie *et al.*, 2003). Any remaining species, given their new position at the sediment/water interface, may be exposed to conditions to which they are not suited. Removal of 30 cm of surface sediment will remove the polychaete and oligochaete community and other species present in the biotope. Recovery of the biological assemblage may take place before the original topography is restored, if the exposed, underlying sediments are similar to those that were removed.

Sensitivity assessment. Extraction of 30 cm of sediment will remove the characterizing biological

component of the biotope. Resistance is assessed as 'None' and biotope resilience is assessed as 'Medium'. Biotope sensitivity is therefore 'Medium'.

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

Abrasion and compaction of the surficial layer may damage individuals, species that live close to the surface or move on the surface such as *Pygospio elegans*, *Corophium volutator* and the lagoon cockle *Cerastoderma glaucum* are likely to be more sensitive than species that burrow deeply into sediments such as *Hediste diversicolor*, oligochaetes and *Arenicola marina*.

No evidence was found to assess the sensitivity of Cerastoderma glaucum but the shells of *Cerastoderma glaucum* are rather thin and brittle, so it is probably quite intolerant of abrasion.

Arenicola marina lives in sediment to a depth of 20-40 cm and is, therefore, protected from most sources of abrasion and physical disturbance caused by surface action. However, it is likely to be damaged by any activity (e.g. anchors, dredging) that penetrates the sediment (see below).

There are few studies on the effects of trampling on sedimentary habitats. Most studies suggest that the effects of trampling across sedimentary habitats depend on the relative proportion of mud to sand (sediment porosity), the dominant infauna (nematodes and polychaetes vs. bivalves) and the presence of burrows (Tyler-Walters & Arnold, 2008). Recovery from impact is relatively fast as shown by Chandrasekara & Frid (1996), where no difference was reported between samples in winter following summer trampling. Wynberg & Branch (1997) suggest that trampling effects are most severe in sediments dominated by animals with stable burrows, as these collapse and the sediment becomes compacted. Rossi *et al.* (2007) examined trampling across intertidal mudflats but were not able to show a significant difference in *Arenicola* abundance between trampled and control sites due to the natural variation in abundance between study sites. Experimental studies on crab-tiling impacts have found that densities of *Tubificoides benedii* and *Tubificoides pseudogaster* were higher in non-trampled plots (Sheehan *et al.*, 2010), indicating that these oligochaetes have some sensitivity to trampling.

Sensitivity assessment. Abrasion may damage or kill a proportion of the population of the characterizing and associated species. *Arenicola marina* and *Tubificoides* spp. that are generally buried more deeply within sediments are likely to be more resistant than species such as *Pygospio elegans* that inhabit fragile tubes that extend above the sediment surface. Biotope resistance is assessed as **Medium** and resilience as **High**, so sensitivity is assessed as **Low**.

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

Disturbance of the sediment below the surface may affect all the characterizing infauna and a number of studies demonstrate effects on the characterizing species.

Mendonça et al. (2008) studied populations of the polychaete *Arenicola marina* at Culbin Sands lagoon, Moray Firth, in NE Scotland. An unprecedented and unexpected cockle harvesting event took place,1.5 years after the start of the sampling programme, which dramatically disturbed the

sediment as it was conducted using tractors with mechanical rakes in some areas, and by boats using a suction dredge in other areas. Therefore, there was an opportunity to compare annual biomass fluctuations "before" and "after" the disturbance. *Arenicola marina* was observed to return to normal activities just a few hours after the disturbance of the sediment during the harvesting event.

Whomersley et al., (2010) conducted experimental raking on intertidal mudflats at two sites (Creeksea- Crouch estuary England and Blackness- lower Forth estuary, Scotland), where *Tubificoides benedii* were dominant species. For each treatment 1 m2 plots were raked twice to a depth of 4cm (using a garden rake). Plots were subject to either low intensity treatments (raking every four weeks) or high (raking every two weeks). The experiment was carried out for 10 months at Creeksea and a year at Blackness. The high and low raking treatments appeared to have little effect on *Tubificoides benedii* (Whomersley et al., 2010).

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

Kaiser *et al.* (2001) carried out experimental hand raking, similar to that used in inter tidal cockle fisheries. Both small and large raked plots showed changed communities in comparison to control plots, smaller plots recovered in 56 days, whilst larger plots remained in an altered state. A three-fold damage rate for undersized cockles was recorded in hand-raked plots compared to control plots.

The extraction of cockles by sediment raking and mechanical disturbance and digging for lugworms causes significant mortality of *Corophium volutator*. Bait digging was found to reduce *Corophium volutator* densities by 39%, juveniles were most affected suffering a 55% reduction in dug areas (Shepherd & Boates, 1999).

Sensitivity assessment. Penetrative gear would probably damage or remove a proportion of the population of *Arenicola marina* and the characterizing oligochaetes but given potential density, the effects may be minor. Other characterizing species including Pygospio elegans and Corophium volutator may be damaged and suffer high levels of mortality. Hence, a resistance of **Low** is suggested. Resilience is probably **Medium**, due to the isolated nature of the sea lochs and lagoons in which this biotope if found, and sensitivity is assessed as **Medium**.

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

This biotope occurs in low energy environments (wave sheltered and weak flow) in shallow isolated water bodies. Deposit feeders are unlikely to be perturbed by increased concentrations of suspended sediment since they live in sediment and are probably adapted to re-suspension of sediment by wave action, during storms or runoff.

In 36-65 day mesocosm studies of the effects of *Arenicola marina* bioturbation, Wendelboe *et al.* (2013) found that the surface of the sediment was dominated by faecal mounds and feeding pits at a flow rate of 0.11 m/s, but was more eroded and the surface was more even at 0.25 m/s. At the low flow (0.11 m/s) there was no change in the sediment. However, at 0.25 m/s, there was a substantial reduction in the silt and clay fractions of the sediment (a 36% reduction) and in the

organic content of the sediment (a 42% reduction). At 0.25 m/s the material ejected into faecal casts was eroded (once the mucilaginous coating had eroded) and the water surface became turbid, resulting in loss of both silt/clay fractions and organic matter.

Sensitivity assessment. The evidence from Wendelboe *et al.* (2013) suggests that an increase in water movement due to storms, or runoff is likely to disturb the sediment surface regularly, especially in winter months, so that the biotope is probably not affected by changes in suspended sediment. In addition, *Arenicola marina* occurs at high abundances in mudflats and sandflats in estuaries where suspended sediment levels may reach grammes per litre. Therefore, a resistance of **High** is suggested so that resilience is **High** (by default) and the biotope is assessed as **Not sensitive** at the benchmark level.

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

Arenicola marina is a sub-surface deposit feeder that derives the sediment it ingests from the surface. It rapidly reworks and mixes sediment. It grows to 12-20 cm in length and lives in burrows to a depth of 20-40 cm. It is unlikely to be perturbed by smothering by 5 cm of sediment. Juveniles may be more susceptible but both adults and juveniles are capable of leaving the sediment and swimming (on the tide) up or down the shore (see Reise *et al.*, 2001). In addition, Gittenberger & Loon (2011) placed *Arenicola marina* into their AMBI Sedimentation Group III, defined as 'species insensitive to higher amounts of sedimentation, but don't easily recover from strong fluctuations in sedimentation'.

Sensitivity assessment. This biotope occurs in a depositional environment, where sedimentation is likely, to be high due to the low energy of the habitat. Therefore, resistance to a deposit of 5 cm of fine sediment is assessed as **High**. Hence, resilience is **High** (by default) and the biotope is probably **Not sensitive** at the benchmark level.

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

Arenicola marina is a sub-surface deposit feeder that derives the sediment it ingests from the surface. It rapidly reworks and mixes sediment. It grows to 12-20 cm in length and lives in burrows to a depth of 20-40 cm. Adults may be able to resist smothering by 30 cm of sediment but juveniles may be more susceptible. Both adults and juveniles are capable of leaving the sediment and swimming (on the tide) up or down the shore (see Reise *et al.*, 2001). In addition, Gittenberger & Loon (2011) placed *Arenicola marina* into their AMBI sedimentation Group III, defined as 'species insensitive to higher amounts of sedimentation, but don't easily recover from strong fluctuations in sedimentation'.

Sensitivity assessment. This biotope occurs in a depositional environment, where sedimentation is likely, to be high due to the low energy of the habitat. However, the deposit of 30 cm in a single event is probably greater than the normal range of sedimentation and, in theses sheltered habitats, likely to remain. Therefore, a proportion of the adults and a greater proportion of the juveniles may not be able to realign themselves with the surface of the sediment and resistance is assessed as **Medium** but at 'Low' confidence due to the lack of direct evidence. Hence, resilience is probably **High** and sensitivity is assessed as **Low** at the benchmark level.

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

DNA damage when exposed to water with toxicants (present in cigarette butts) in quantities 60 fold lower than reported from urban run-off (Wright *et al.*, 2015). Studies are limited on impacts of litter on infauna and this UK study suggests health of infauna populations are negatively impacted by this pressure.

Studies of sediment dwelling, sub surface deposit feeding worms, a trait shared by species abundant in this biotope, showed negative impacts from ingestion of micro plastics. For instance, *Arenicola marina* ingests micro-plastics that are present within the sediment it feeds within. Wright *et al.* (2013) carried out a lab study that displayed presence of micro-plastics (5% UPVC) significantly reduced feeding activity when compared to concentrations of 1% UPVC and controls. As a result, *Arenicola marina* showed significantly decreased energy reserves (by 50%), took longer to digest food, and decreased bioturbation levels. These effects would be likely to impact colonisation of sediment by other species, reducing diversity in the biotopes the species occurs within. Wright *et al.* (2013) also present a case study based on their results, that in the intertidal regions of the Wadden Sea, where *Arenicola marina* is an important ecosystem engineer, *Arenicola marina* could ingest 33m; of micro plastics a year.

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

Sensitivity assessment. Impacts from the pressure 'litter' would depend upon the exact form of litter and the extent. In the case of marine litter in the form of cigarette butts or micro plastics health of populations of characterizing species would be impacted. Significant impacts have been shown in laboratory studies but impacts at biotope scales are still unknown and this pressure is 'Not assessed'.

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

No evidence was found

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

Species within the biotope can probably detect vibrations caused by noise and in response may retreat into the sediment for protection. However, at the benchmark level, the community is

unlikely to be respond to noise and therefore is **Not sensitive.**

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

All characterizing species live in the sediment and do not rely on light levels directly to feed or so limited direct impact is expected. As this biotope is not characterized by the presence of primary producers it is not considered that shading would alter the character of the habitat directly.

Beneath shading structures, there may be changes in microphytobenthos abundance. This biotope may support microphytobenthos on the sediment surface and within the sediment. Mucilaginous secretions produced by these algae may stabilise fine substrata (Tait & Dipper, 1998). Shading will prevent photosynthesis leading to death or migration of sediment microalgae altering sediment cohesion and food supply to deposit feeders like *Arenicola* and synaptid holothurians, although they fed on a range of organic matter within the sediment.

Sensitivity assessment. Therefore, biotope resistance is assessed as '**High**' and resilience is assessed as '**High**' (by default) and the biotope is considered to be '**Not sensitive'**.

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

Not relevant - this pressure is considered applicable to mobile species, e.g. fish and marine mammals rather than seabed habitats. Physical and hydrographic barriers may limit the dispersal of seed. But seed dispersal is not considered under the pressure definition and benchmark.

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

Not relevant to seabed habitats.

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

Not relevant.

🐚 Biological Pressures

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

Important characterizing species within this biotope are not genetically modified or translocated. Therefore, this pressure is considered 'Not relevant' to this biotope group.

Introduction or spread of invasive non-indigenous species

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

No evidence.

Introduction of microbial pathogens

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

Ashworth (1904) recorded the presence of distomid cercariae and Coccidia in *Arenicola marina* from the Lancashire coast. However, no information concerning infestation or disease related mortalities was found.

Bacterial diseases are more significant in the larval stages and protozoans are the most common cause of epizootic outbreaks leading to mass mortalities of bivalves. Parasitic worms, trematodes, cestodes and nematodes can reduce growth and fecundity in bivalves and may in some instances cause death (Dame, 1996).

In the Danish Wadden Sea, 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. This event coincided with a high positive NAO index, high temperatures, strong winds, and increased precipitation in northern Europe (Mouritsen & Poulin, 2002a, and references therein). The transmission rates of larval parasites from snail to amphipods and the rate of parasite-induced amphipod mortality are both strong positive functions of temperature (Jensen & Mouritsen, 1992; Mouritsen, 2002). 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.

Marine oligochaetes host numerous protozoan parasites without apparent pathogenic effects even at high infestation levels (Giere & Pfannkuche, 1982 and references therein)

Sensitivity assessment. Outbreaks of pathogens and parasites may occur in this biotope and lead to changes in species abundance and composition. Biotope resistance is assessed as 'Medium' as the evidence suggests that infections are species-specific and changes in abundance are unlikely to result in biotope classification. Resilience is assessed as 'High' and biotope sensitivity is assessed as 'Low'.

Removal of target species

Low	Medium	Medium
Q: High A: Medium C: Medium	Q: High A: High C: High	Q: High A: Medium C: Medium

Within this biotope the polychaetes *Arenicola marina* and *Hediste diversicolor* may be targeted recreationally or commercially. The lagoon cockle *Cerastoderma glaucum* is targeted in some parts of its range and may be targeted but no evidence was found.

Smaller scale extraction of patches of substratum through activities such as bait digging for *Hediste diversicolor* may have impacts over finer spatial scales within the biotope. If the impact is not spread over a larger area the effects are likely to occur within the dug area. However, Neves de

Carvalho *et al.* (2013) calculated that bait digging activities in the Douro estuary, Portugal may remove up to 9.9 tonnes of *Hediste diversicolor*.

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 re-populated within 1 month, whereas mounds of dug sediment took showed a reduced population. The basins accumulated fine sediment and organic matter and showed increased population levels for about 2-3 months after digging.

Fowler (1999) reviewed the effects of bait digging on intertidal fauna, including *Arenicola marina*. Diggers were 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⬜. 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 were used in the Dutch Wadden Sea where they removed 17-20 million lugworms/year. However, when combined with hand digging the harvest represented only 0.75% of the estimated population in the area. 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 are 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 reached its original level in at least three years.

Corophium species are affected by sediment disturbance from deposit feeders and removal of *Arenicola marina* and *Cerastoderma glaucum* may be beneficial for these species. The sediment turnover caused by cockles and lugworms disturbs the burrows of *Corophium volutator* and caused a significant negative effect on *Corophium volutator* density as a result of increased rate of swimming making the amphipod more vulnerable to predation (Flach & de Bruin, 1993, 1994). *Cerastoderma edule* can entrain large amounts of settling bivalve (40% -André & Rosenberg, 1991): removal of *Cerastoderma glaucum* may enhance recruitment of other species as it is likely to also consumes larvae. *Corophium* species are affected by sediment disturbance from deposit feeders (Flach, 1992) and removal of *Arenicola marina* and *Cerastoderma glaucum* may be beneficial for these species. The activities of *Arenicola marina* also reduce the abundance of *Pygospio elegans* (Wilson, 1981).

Sensitivity assessment. Recreational bait digging may remove a proportion of the population of *Hediste diversicolor* and *Arenicola*. The physical effects of removal are addressed under penetration above. *Arenicola marina* is a bioturbator and ecosystem engineer and its removal would probably have a significant effect on the nature of the sediment and the other species that could inhabit the sediment. As both species are relatively sedentary and targeted removal is likely to remove large proportions of the population, a resistance of **Low** is suggested. Resilience is probably **Medium,** due to the isolated nature of the sea lochs and lagoons in which this biotope if found, and sensitivity is assessed as **Medium**. This biotope occurs in the sublittoral fringe and this may limit targeted removal where the biotope is never exposed during the tidal cycle.

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

Incidental removal of the characterizing species would alter the character of the biotope and the delivery of ecosystem services such as secondary production and bioturbation. *Arenicola marina* is

a bioturbator and ecosystem engineer and its incidental removal would probably have a significant effect on the nature of the sediment and the other species that could inhabit the sediment.

Many lagoons are important feeding areas for birds (Barnes, 1994) and removal of species such as *Hydrobia ulvae*, *Cerastodoerma glaucum* and Corophium spp. will alter food supply. Fish may also feed on bivalve siphons and tails of *Arenicola marina* (Barnes, 1994). The removal of *Chaetomorpha linum* and other vegetation will remove refugia enhancing predation on species (Barnes, 1994). Juvenile *Cerastoderma glaucum* inhabit *Chaetomorpha linum* before settling on sediments (Ivell, 1979), removal of *Chaetomorpha linum* will therefore remove habitat for juveniles. Populations of oligochaetes provide food for macroinvertebrates fish and birds. For example up to 67% of flounder and plaice stomachs examined from the Medway estuary (UK) (Van den Broek, 1978) contained the remains of *Tubificoides benedii (*studied as *Peloscolex benedeni)* and shrimps which in turn support higher trophic levels (predatory birds and fish). For some migratory birds the characterizing species *Tubificoides benedii* can form an important part of the diet during winter (Bagheri & McLusky, 1984). Polychaetes and crustaceans are also predators of oligochaetes and may significantly reduce numbers (Giere & Pfannkuche, 1982 and references therein). The loss of the oligochaete population could, therefore, impact other trophic levels.

Sensitivity assessment. Removal of the characterizing species would alter the character of the biotope. Resistance is therefore assessed as 'Low' and resilience as 'Medium' so that sensitivity is categorised as 'Low'.

Bibliography

Anderson, A., 1971. Intertidal activity, breeding and the floating habit of *Hydrobia ulvae* in the Ythan estuary. *Journal of the Marine Biological Association of the United Kingdom*, **51**, 423-437.

André, C. & Rosenberg, R., 1991. Adult-larval interactions in the suspension-feeding bivalves *Cerastoderma edule* and *Mya arenaria*. *Marine Ecology Progress Series*, **71** (3), 227-234.

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.

Ashworth, J.H., 1904. Arenicola *(the lug-worm)*. Liverpool: Liverpool Marine Biology Committee. [L.M.B.C. Memoirs XI].

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

Bamber, R., Gilliland, P. & Shardlow, M., 2001. Saline lagoons: A guide to their management and creation. *English Nature, Peterborough*, 171 pp.

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

Barnes, R.S.K., 1980b. *Coastal lagoons. The natural history of a neglected habitat*. Cambridge: Cambridge University Press.

Barnes, R.S.K., 1988. On reproductive strategies in adjacent lagoonal and intertidal marine populations of the gastropod *Hydrobia ulvae*. *Journal of the Marine Biological Association of the United Kingdom*, **68**, 365-375.

Barnes, R.S.K., 1990. Reproductive strategies in the contrasting populations of the coastal gastropod *Hydrobia ulvae*. II. Longevity and lifetime egg production. *Journal of Experimental Marine Biology and Ecology*, **138**, 183-200.

Barnes, R.S.K., 1991. Dilemmas in the theory and practice of biological conservation as exemplified by British coastal lagoons. *Biological Conservation*, 55, 315 - 323.

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

Bat, L. & Raffaelli, D., 1998. Sediment toxicity testing: a bioassay approach using the amphipod *Corophium volutator* and the polychaete *Arenicola marina*. *Journal of Experimental Marine Biology and Ecology*, **226**, 217-239.

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.

Bentley, M.G. & Pacey, A.A., 1992. Physiological and environmental control of reproduction in polychaetes. *Oceanography and Marine Biology: an Annual Review*, **30**, 443-481.

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. & Dekker, R., 2005. Decline of recruitment success in cockles and other bivalves in the Wadden Sea: possible role of climate change, predation on postlarvae and fisheries. *Marine Ecology Progress Series*, **287**, 149-167.

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.

Binyon, J., 1966. Salinity tolerance and ionic regulation. In *Physiology of Echinodermata* (ed. R.A. Boolootian), pp. 359-377. New York: John Wiley & Sons.

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

Boalch, G.T., 1979. The dinoflagellate bloom on the coast of south-west England, August to September 1978. *Journal of the Marine Biological Association of the United Kingdom*, **59**, 515-517.

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

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.

Boyden, C.R., 1972. Behaviour, survival and respiration of the cockles *Cerastoderma edule* and *C. glaucum* in air. *Journal of the Marine Biological Association of the United Kingdom*, **52**, 661-680.

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, A.E., Burn, A.J., Hopkins, J.J. & Way, S.F., 1997. The habitats directive: selection of Special Areas of Conservation in the UK. *Joint Nature Conservation Committee, Peterborough, JNCC Report* no. 270.

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.

Browne, M.A., Underwood, A.J., Chapman, M.G., Williams, R., Thompson, R.C. & van Franeker, J.A., 2015. Linking effects of

anthropogenic debris to ecological impacts. Proceedings of the Royal Society B: Biological Sciences, 282 (1807), 20142929.

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

Bryan, G.W., 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., 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.

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

Clay, E., 1960. Literature survey of the common fauna of estuaries. 8. *Hydrobia ulvae* (Pennant), *Hydrobia ventrosa* (Montagu) and *Potamopyrgus jenkinsi* (Smith). *Imperial Chemical Industries Ltd, Paints Division, Research Department Memorandum* PVM45/B/483.

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/

Commito, J.A. & Boncavage, E.M., 1989. Suspension-feeders and coexisting infauna: an enhancement counterexample. *Journal of Experimental Marine Biology and Ecology*, **125** (1), 33-42.

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.

Conti, E., 1987. Acute toxicity of three detergents and two insecticides in the lugworm, *Arenicola marina* (L.): a histological and a scanning electron microscopic study. *Aquatic toxicology*, **10** (5-6), 325-334.

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.

Cromey, C., Black, K., Edwards, A. & Jack, I., 1998. Modelling the deposition and biological effects of organic carbon from marine sewage discharges. *Estuarine, Coastal and Shelf Science*, **47** (3), 295-308.

Cryer, M., Whittle, B.N. & Williams, K., 1987. The impact of bait collection by anglers on marine intertidal invertebrates. *Biological Conservation*, **42**, 83-93.

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

Dame, R.F.D., 1996. *Ecology of Marine Bivalves: an Ecosystem Approach.* New York: CRC Press Inc. [Marine Science Series.]

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

De Wilde P.A.W.J. & Berghuis, E.M., 1979. Laboratory experiments on growth of juvenile lugworms, *Arenicola marina*. *Netherlands Journal of Sea Research*, **13**, 487-502.

Dernie, K.M., Kaiser, M.J., Richardson, E.A. & Warwick, R.M., 2003. Recovery of soft sediment communities and habitats following physical disturbance. *Journal of Experimental Marine Biology and Ecology*, **285-286**, 415-434.

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

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

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

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

Ducrotoy, C.R., Rybarczyk, H., Souprayen, J., Bachelet, G., Beukema, J.J., Desprez, M., Dörjes, J., Essink, K., Guillou, J., Michaelis, H., Sylvand, B., Wilson, J.G., Elkaïm, B. & Ibanez, F., 1991. A comparison of the population dynamics of the cockle (*Cerastoderma edule*) in North-Western Europe. In *Proceedings of the Estuarine and Coastal Sciences Association Symposium, ECSA 19, 4-8 September 1989, University of Caen, France. Estuaries and Coasts: Spatial and Temporal Intercomparisons*, pp. 173-184. Denmark: Olsen & Olsen.

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.*

Elliott, M., 1998. Summary of effects of commercial fisheries on estuarine ecosystems: a European perspective. , *Unpublished report to SCOR working group 105, Halifax NS, March 1998. P57.*

Farke, H. & Berghuis, E.M., 1979. Spawning, larval development and migration behaviour of *Arenicola marina* in the laboratory. *Netherlands Journal of Sea Research*, **13**, 512-528.

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. & 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. & 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.

Flach, E.C., 1992. Disturbance of benthic infauna by sediment-reworking activities of the lugworm *Arenicola marina*. *Netherlands Journal of Sea Research*, **30**, 81-89.

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., 1994. *British prosobranch molluscs: their functional anatomy and ecology*, revised and updated edition. London: The Ray Society.

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

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

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

Gibson, G.D. & Harvey, J., 2000. Morphogenesis during asexual reproduction in *Pygospio elegans* Claparede (Annelida, Polychaeta). *The Biological Bulletin*, **199** (1), 41-49.

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

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

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

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

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

Gorbushin, A.M., Levakin, I.A., Panchina, N.A. & Panchin, Y.V., 2001. *Hydrobia ulvae* (Gastropoda: Prosobranchia): a new model for regeneration studies. *Journal of Experimental Biology*, **204** (2), 283-289.

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

Gray, J.S., Wu R.S.-S. & Or Y.Y., 2002. Effects of hypoxia and organic enrichment on the coastal marine environment. *Marine Ecology Progress Series*, **238**, 249-279.

Günther, C-P., 1992. Dispersal of intertidal invertebrates: a strategy to react to disturbances of different scales? *Netherlands Journal of Sea Research*, **30**, 45-56.

Hall, J.A. & Frid, C.L.J., 1995. Response of estuarine benthic macrofauna in copper-contaminated sediments to remediation of sediment quality. *Marine Pollution Bulletin*, **30**, 694-700.

Hannam, M.L., Hagger, J.A., Jones, M.B. & Galloway, T.S., 2008. Characterisation of esterases as potential biomarkers of pesticide exposure in the lugworm *Arenicola marina* (Annelida : Polychaeta). *Environmental Pollution*, **152** (2), 342-350.

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.

Haskoning UK Ltd. 2006. Investigation into the impact of marine fish farm deposition on maerl beds. *Scottish Natural Heritage*

Commissioned Report No. 213 (ROAME No. AHLA10020348).

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

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.

Holt, T.J., Jones, D.R., Hawkins, S.J. & Hartnoll, R.G., 1995. The sensitivity of marine communities to man induced change - a scoping report. *Countryside Council for Wales, Bangor, Contract Science Report*, no. 65.

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

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

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

Hyman, L.V., 1955. *The Invertebrates:* Vol. IV. *Echinodermata. The coelomate Bilateria.* New York: McGraw Hill.

Ivell, R., 1979. The biology and ecology of a brackish lagoon bivalve, *Cerastoderma glaucum* Bruguiere, in an English lagoon, the Widewater, Sussex. *Journal of Molluscan Studies*, **45**, 383-400.

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

Johnston, R., 1984. Oil Pollution and its management. 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.1433-1582. New York: John Wiley & Sons Ltd.

Kaiser, M., Broad, G. & Hall, S., 2001. Disturbance of intertidal soft-sediment benthic communities by cockle hand raking. *Journal of Sea Research*, **45** (2), 119-130.

Kennedy, V.H., Horrill, A.D. & Livens, F.R., 1988. Radioactivity and wildlife. *Institute of Terrestrial Ecology, NCC/NERC Contract HF 3-08-21 (10). TFS Project T07006GL.*, Merlewood Research Station.

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.

Lindsay, S.M., Jackson, J.L. & He, S.Q., 2007. Anterior regeneration in the spionid polychaetes *Dipolydora quadrilobata* and *Pygospio elegans. Marine Biology*, **150** (6), 1161-1172.

Linke, O., 1939. Die Biota des Jadebusenwatts. *Helgolander Wissenschaftliche Meeresuntersuchungen*, **1**, 201-348.

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

Longbottom, M.R., 1970. The distribution of *Arenicola marina* (L.) with particular reference to the effects of particle size and organic matter of the sediments. *Journal of Experimental Marine Biology and Ecology*, **5**, 138-157.

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

McLusky, D.S., 1989. *The Estuarine Ecosystem*, 2nd ed. New York: Chapman & Hall.

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

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

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

Mendonça, V.M., Raffaelli, D.G., Boyle, P. & Hoskins, S., 2008. Spatial and temporal characteristics of benthic invertebrate communities at Culbin Sands lagoon, Moray Firth, NE Scotland, and impact of the disturbance of cockle harvesting. *Scientia Marina*, **72** (2), 265-278.

Mouritsen, K.N. & Poulin, R., 2002. Parasitism, community structure and biodiversity in intertidal ecosystems. *Parasitology*, **124**, 101-117.

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.

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, G.E., 1948. A contribution to our knowledge of the life history of *Arenicola marina* L. *Journal of the Marine Biological Association of the United Kingdom*, **28**, 554-580.

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.

Nilsson, H.C. & Rosenberg, R., 1994. Hypoxic response of two marine benthic communities. *Marine Ecology Progress Series*, **115**, 209-217.

OBIS, 2016. Ocean Biogeographic Information System (OBIS). http://www.iobis.org, 2016-03-15

Olive, P.J.W. & Cadman, P.S., 1990. Mass mortalities of the lugworm on the South Wales coast: a consequence of algal bloom? *Marine Pollution Bulletin*, **21**, 542-545.

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

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.

Pilkington, M.C., 1971. The veliger stage of *Hydrobia ulvae* (Pennant). *Proceedings of the Malacological Society of London*, **39**, 281-287.

Prouse, N.J. & Gordon, D.C., 1976. Interactions between the deposit feeding polychaete *Arenicola marina* and oiled sediment. In *Proceedings of a Symposium of the American Institute of Biological Sciences, Arlington, Virginia, 1976. Sources, effects and sinks of hydrocarbons in the aquatic environment*, pp. 408-422. USA: American Institute of Biological Sciences.

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.

Rankin, C.J. & Davenport, J.A., 1981. *Animal Osmoregulation*. Glasgow & London: Blackie. [Tertiary Level Biology].

Reise, K., Simon, M. & Herre, E., 2001. Density-dependent recruitment after winter disturbance on tidal flats by the lugworm *Arenicola marina*. *Helgoland Marine Research*, **55**(3), 161-165.

Retraubun, A.S.W., Dawson, M. & Evans, S.M., 1996. Spatial and temporal factors affecting sediment turnover by the lugworm *Arenicola marina* (L). *Journal of Experimental Marine Biology and Ecology*, **201** (1-2), 23-35.

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.

Rossi, F., Forster, R., Montserrat, F., Ponti, M., Terlizzi, A., Ysebaert, T. & Middelburg, J., 2007. Human trampling as short-term disturbance on intertidal mudflats: effects on macrofauna biodiversity and population dynamics of bivalves. *Marine Biology*, **151** (6), 2077-2090.

Schroeer, M., Wittmann, A.C., Gruener, N., Steeger, H.-U., Bock, C., Paul, R. & Poertner, H.-O., 2009. Oxygen limited thermal tolerance and performance in the lugworm *Arenicola marina*: A latitudinal comparison. *Journal of Experimental Marine Biology and Ecology*, **372** (1-2), 22-30.

Sheehan, E., Coleman, R., Thompson, R. & Attrill, M., 2010. Crab-tiling reduces the diversity of estuarine infauna. *Marine Ecology Progress Series*, **411**, 137-148.

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.

Shumway, S.E. & Davenport, J., 1977. Some aspects of the physiology of *Arenicola marina* (Polychaeta) exposed to fluctuating salinities. *Journal of the Marine Biological Association of the United Kingdom*, **57**, 907-924.

Smith, J.E. (ed.), 1968. 'Torrey Canyon'. *Pollution and marine life.* Cambridge: Cambridge University Press.

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

Snelgrove, P.V., Butman, C.A. & Grassle, J.F., 1995. Potential flow artifacts associated with benthic experimental gear: deep-sea mudbox examples. *Journal of Marine Research*, **53** (5), 821-845.

Sola, J.C., 1996. Population dynamics, reproduction and growth and secondary production of the mud snail *Hydrobia ulvae* (Pennant) *Journal of Experimental Marine Biology and Ecology*, **205**, 49-62.

Sommer, A. & Portner, H.O., 1999. Exposure of *Arenicola marina* to extreme temperatures: adaptive flexibility of a boreal and a subpolar population. *Marine Ecology Progress Series*, **181**, 215-226.

Sommer, A., Klein, B. & Pörtner, H.O., 1997. Temperature induced anaerobiosis in two population of the polychaete worm

Arenicola marina (L.). *Journal of Comparative Physiology*, series B, **167**, 25-35.

Spencer, T. & Brooks, S., 2012. Methodologies for measuring and modelling change in coastal saline lagoons under historic and accelerated sea-level rise, Suffolk coast, eastern England. *Hydrobiologia*, **693** (1), 99-115.

Stickle, W.B. & Diehl, W.J., 1987. Effects of salinity on echinoderms. In *Echinoderm Studies, Vol. 2* (ed. M. Jangoux & J.M. Lawrence), pp. 235-285. A.A. Balkema: Rotterdam.

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

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

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

Thorp, K., Dalkin, M., Fortune, F. & Nichols, D., 1998. *Marine Nature Conservation Review, Sector 14. Lagoons in the Outer Hebrides: area summaries.* Peterborough: Joint Nature Conservation Committee. [Coasts and seas of the United Kingdom. MNCR Series.]

Thorpe, K., 1998. *Marine Nature Conservation Review, Sectors 1 and 2. Lagoons in Shetland and Orkney.* Peterborough: Joint Nature Conservation Committee. [Coasts and seas of the United Kingdom. MNCR Series.]

Trueman, E.R. & Ansell, A.D., 1969. The mechanisms of burrowing into soft substrata by marine animals. *Oceanography and Marine Biology: an Annual Review*, **7**, 315-366.

Tyler-Walters, H. & Arnold, C., 2008. Sensitivity of Intertidal Benthic Habitats to Impacts Caused by Access to Fishing Grounds. *Report to Cyngor Cefn Gwlad Cymru / Countryside Council for Wales from the Marine Life Information Network (MarLIN) [Contract no. FC 73-03-327]*, Marine Biological Association of the UK, Plymouth, 48 pp. Available from: www.marlin.ac.uk/publications

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

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

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

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

Watson, G.J., Williams, M.E. & Bentley, M.G., 2000. Can synchronous spawning be predicted from environmental parameters? A case study of the lugworm *Arenicola marina*. *Marine Biology*, **136** (6), 1003-1017.

Wells, G.P., 1945. The mode of life of *Arenicola marina* L. *Journal of the Marine Biological Association of the United Kingdom*, **26**, 170-207.

Wendelboe, K., Egelund, J.T., Flindt, M.R. & Valdemarsen, T., 2013. Impact of lugworms (*Arenicola marina*) on mobilization and transport of fine particles and organic matter in marine sediments. *Journal of Sea Research*, **76**, 31-38.

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

Wilde de P.A.W.J. & Berghuis, E.M., 1979. Spawning and gamete production inn *Arenicola marina* in the Netherlands, Wadden Sea. *Netherlands Journal of Sea Research*, **13**, 503-511.

Wilson, J.G., 1981. Temperature tolerance of circatidal bivalves in relation to their distribution. *Journal of Thermal Biology*, **6**, 279-286.

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.

Wright, E.P., Kemp, K., Rogers, A.D. & Yesson, C., 2015. Genetic structure of the tall sea pen *Funiculina quadrangularis* in NW Scottish sea lochs. *Marine Ecology*, **36** (3), 659-667.

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

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

Wynberg, R.P. & Branch, G.M., 1997. Trampling associated with bait-collection for sandprawns *Callianassa kraussi* Stebbing: Effects on the biota of an intertidal sandflat. *Environmental Conservation*, **24** (2), 139-148.

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