Sparse fauna (barnacles and spirorbids) on sand/pebble-scoured rock in littoral caves

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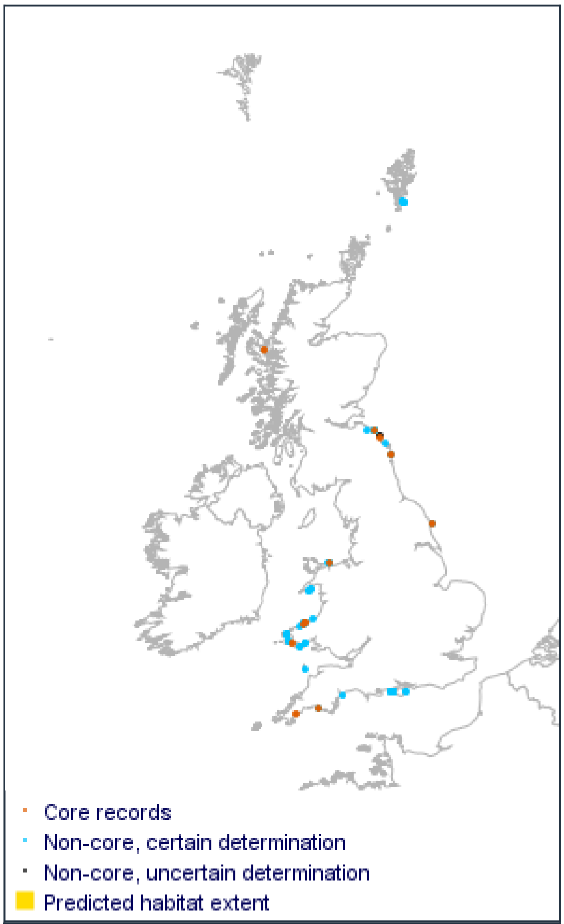

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Biotope distribution data provided by EMODnet Seabed Habitats (www.emodnet-seabedhabitats.eu)

Researched by Dr Heidi Tillin **Refereed by** Admin

Summary

UK and Ireland classification

EUNIS 2008	A1.449	Sparse fauna (barnacles and spirorbids) on sand/pebble-scoured rock in littoral caves
JNCC 2015	LR.FLR.CvOv.ScrFa	Sparse fauna (barnacles and spirorbids) on sand/pebble-scoured rock in littoral caves
JNCC 2004	LR.FLR.CvOv.ScrFa	Sparse fauna (barnacles and spirorbids) on sand/pebble-scoured rock in littoral caves
1997 Biotope		

Description

Upper to lower shore sand- or pebble-scoured cave walls characterized by an impoverished faunal assemblage which may include bryozoan crusts, scattered sponges *Halichondria panicea*, barnacles such as *Semibalanus balanoides* or often large *Balanus crenatus and* the limpet *Patella vulgata*. The isopod *Ligia oceanica* may seek refuge in crevices in the rock, and due to the decreased effect of desiccation in these damp caves, other species such as the anemone *Actinia equina* and spirorbid polychaetes are able to extend further up the shore than normally found on open rock. The lower section of the wall which is subject to greatest scour may be characterized by a band of *Spirobranchus triqueter* and spirorbid tube-forming polychaetes. In wave sheltered conditions, this biotope may extend to the cave ceiling. The rear of caves on the lower shore may support only sparse fauna consisting of spirorbid polychaetes and barnacles such as *Chthamalus montagui* with scattered *Spirobranchus* sp., scattered bryozoan and coralline crusts and in the south-west, occasional *Sabellaria alveolata*. Shade-tolerant red algae such as *Lomentaria articulata* may occasionally occur. Due to the low species abundance in this biotope, there may be a variation from cave to cave, depending on local conditions (JNCC, 2015).

Depth range

Mid shore

Additional information

-

Listed By

- none -

Further information sources

Search on:

JNCC

Sensitivity review

Sensitivity characteristics of the habitat and relevant characteristic species

The biotope description and characterizing species is taken from JNCC (2015). This biotope is characterized by an impoverished faunal assemblage that typically includes barnacles such as *Chthamalus* spp., *Semibalanus balanoides* or *Balanus crenatus*. Where scour is greatest the biotope may be characterized by a band of *Spirobranchus* (previously *Pomatoceros*) *triqueter* and spirorbid tube-forming polychaetes. As the barnacles and spirorbids are named in the biotope description they are considered key defining species and the sensitivity assessments are focussed on these species. oralline crusts are typically present in this biotope and their sensitivity is assessed. Although other species may be associated with this biotope, such as shade-tolerant red algae, limpets, isopods, anemones and sponges, these are present in low abundances and are not considered to be characterizing and are therefore considered only generally within the assessments.

Abrasion resulting from scouring by sand and pebbles is a key factor structuring the biotope and significant alteration to scouring is likely to change the character of the biotope. Where pressures may alter this factor, this is identified and discussed within the sensitivity assessments.

Resilience and recovery rates of habitat

This biotope is subject to high levels of physical disturbance from abrasion. The species that characterize it are therefore robust animals that can withstand some physical disturbance and/or recover rapidly. This biotope is therefore considered to have a high recovery potential. Sponges and anemones can repair damage and regenerate from small, surviving body parts. Other species such as limpets and ispods are mobile and can migrate into the biotope as adults, while other attached species such as the barnacles and spirorbids produce large numbers of pelagic larvae that can recolonize suitable habitats. Most of the epifauna is probably subject to severe physical disturbance and scour during winter storms and probably develops annually, through regrowth, recolonization and migration from adjacent habitats. Therefore, recovery is likely to be rapid as a typical biological assemblage develops within less than year and probably within 6 months in spring and summer.

Spirorbids are rapid colonizers but poor competitors and hence are maintained in this biotope by the high levels of physical disturbance. Recovery may be within as little as 3 months for these species, based on rapid settlement on artificial panels (Saunders & Connell, 2001; James & Underwood, 1994).Sebens (1985, 1986) observed that calcareous tube worms, encrusting bryozoans and erect hydroids and bryozoans covered scraped areas within 4 months in spring, summer and autumn. Similarly, Hiscock (1983) noted that a community, under conditions of scour and abrasion from stones and boulders moved by storms, developed into a community similar to this biotope, consisting of fast growing species such as *Spirobranchus* (formerly *Pomatoceros*) *triqueter*. Off Chesil Bank, the epifaunal community dominated by *Spirobranchus* (as *Pomatoceros) triqueter* and *Balanus crenatus* decreased in cover in October as it was scoured away in winter storms, but recolonized in May to June (Gorzula 1977). Warner (1985) reported that the community did not contain any persistent individuals but that recruitment was sufficiently predictable to result in a dynamic stability and a similar community, dominated by *Spirobranchus* (as *Pomatoceros) triqueter*, *Balanus crenatus* and *Electra pilosa*, (an encrusting bryozoan), was present in 1979, 1980 and 1983 (Riley and Ballerstedt, 2005).

On rocky shores, barnacles are often quick to colonize available gaps, although a range of factors, as outlined below, will influence whether there is a successful episode of recruitment in a year to re-populate a shore following impacts. Bennell (1981) observed that barnacles that were removed when the surface rock was scraped off in a barge accident at Amlwch, North Wales returned to pre-accident levels within 3 years. Petraitis & Dudgeon (2005) also found that *Semibalanus balanoides* quickly recruited (present a year after and increasing in density) to experimentally cleared areas within the Gulf of Maine, that had previously been dominated by *Ascophyllum nodosum* However, barnacle densities were fairly low (on average 7.6 % cover) as predation levels in smaller patches were high and heat stress in large areas may have killed a number of individuals (Petraitis *et al*., 2003). Following creation of a new shore in the Moray Firth, *Semibalanus balanoides* did not recruit in large numbers until 4 years after shore creation (Terry & Sell, 1986).

Successful recruitment of high number of *Semibalanus balanoides* individuals to replenish the population may be episodic (Kendall *et al*., 1985). After settlement the juveniles are subject to high levels of predation as well as dislodgement from waves and sand abrasion depending on the area of settlement. *Semibalanus balanoides* may live up to 4 years in higher areas of the shore (Wethey,1985). Predation rates are variable (see Petraitis et al., 2003) and are influenced by a number of factors including the presence of algae (that shelters predators such as the dog whelk, Nucella lapillus, and the shore crab, *Carcinus maenas* and the sizes of clearings (as predation pressure is higher near canopies (Petraitis *et al*., 2003). Local environmental conditions, including surface roughness (Hills & Thomason, 1998), wind direction (Barnes, 1956), shore height, wave exposure (Bertness *et al*., 1991) and tidal currents (Leonard *et al*., 1998) have been identified, among other factors, as factors affecting settlement of *Semibalanus balanoides*. Biological factors such as larval supply, competition for space, presence of adult barnacles (Prendergast *et al*., 2009) and the presence of species that facilitate or inhibit settlement (Kendall, *et al*., 1985, Jenkins *et al*., 1999) also play a role in recruitment. Mortality of juveniles can be high but highly variable, with up to 90 % of *Semibalanus balanoide*s dying within ten days (Kendall *et al*., 1985). Presumably these factors also influence the transport, supply and settlement of *Chthamaus* spp., *Balanus crenatus* and other species such as spirorbids that produce pelagic larvae.

Resilience assessment. Resilience is assessed as 'High' (within 2 years) for all levels of impact, even where resistance is none, as it is likely that a similar community can rapidly develop.

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 recognisable as the initial habitat of interest. It should be noted that the recovery rates are only indicative of the recovery potential.

Hydrological Pressures

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

Examples of distribution and thermal tolerances tested in laboratory experiments are provided as evidence to support the sensitivity assessment. In general, populations can acclimate to prevailing conditions which can alter tolerance thresholds and care should, therefore, be used when interpreting reported tolerances. Species that are found in the intertidal are exposed to extremes of high and low air temperatures during periods of emersion. They also experience temperature fluctuation over a short period of time during the tidal cycle. In winter air temperatures are colder than the sea, conversely in summer air temperatures are much warmer than the sea. In general intertidal species are therefore able to tolerate a wide range of temperatures. Within this biotope the cave habitat provides some shade and hence cooler temperatures and reduced desiccation supporting species typically found lower on the shore such as *Balanus crenatus* and encrusting corallines.

Spirorbids are distributed circumglobally, with some differences in range between taxonomic groups (Knight-Jones *et al*., 1991). the biotope classification does not specify species or families and therefore it is considered that a spirorbid group may be able to colonize this biotope regardless of temperature changes.

The barnacles *Semibalanus balanoides* and *Balanus crenatus* are both ‘northern species’. *Semibalanus balanoides* extend from Portugal or Northern Spain to the Arctic circle. Populations in the southern part of England are therefore relatively close to the southern edge of their geographic range. *Semibalanus balanoides* are found on the mid-shore but are less resistant to desiccation that the 'southern' Chthamalus barnacle species. Long-term time series show that successful recruitment of *Semibalanus balanoides* is correlated to sea temperatures (Mieszkowska, *et al*., 2014) and that due to recent warming its range has been contracting northwards. Temperatures above 10 to 12 °C inhibit reproduction (Barnes, 1957, 1963, Crisp & Patel, 1969) and laboratory studies suggest that temperatures at or below 10 °C for 4-6 weeks are required in winter for reproduction, although the precise threshold temperatures for reproduction are not clear (Rognstad *et al*., 2014). Observations of recruitment success in *Semibalanus balanoides* throughout the South West of England, strongly support the hypothesis that an extended period (4-6 weeks) of sea temperatures <10 °C is required to ensure a good supply of larvae (Rognstad *et al*., 2014, Jenkins *et al*., 2000). During periods of high reproductive success, linked to cooler temperatures, the range of barnacles has been observed to increase, with range extensions in the order of 25 km (Wethey *et al*., 2011), and 100 km (Rognstad *et al*., 2014).

Balanus crenatus is described as a boreal species (Newman & Ross, 1976) it is found throughout the northeast Atlantic from the Arctic to the west coast of France as far south as Bordeaux; east and west coasts of North America and Japan. In Queens Dock, Swansea where the water was on average 10°C higher than average due to the effects of a condenser effluent, *Balanus crenatus* was replaced by the subtropical barnacle *Balanus amphitrite*. After the water temperature cooled *Balanus crenatus* returned (Naylor, 1965). The increased water temperature in Queens Dock is greater than an increase at the pressure benchmark (2-5°C). *Balanus crenatus* has a peak rate of cirral beating at 20°C and all spontaneous activity ceases at about 25°C (Southward, 1955). The tolerance of *Balanus crenatus*, collected in the summer (and thus acclimated to higher temperatures), to increased temperatures was tested in the laboratory. The median upper lethal temperature tolerance was -25.2°C (Davenport & Davenport, 2005) confirming the observations of Southward (1955).

Increased temperatures are likely to favour Chthamalid barnacles present in the biotope rather than *Semibalanus balanoides* (Southward *et al*. 1995) and *Balanus crenatus*. *Chthamalus montagui* and *Chthamalus stellatus* are warm water species, with a northern limit of distribution in Britain so are likely to be tolerant of long-term increases in temperature. The range of *Chthamalus stellatus* and *Chthamalus montagui* has been extending northwards due to increasing temperatures. *Chthamalus* suffers failure of fertilization at temperatures of 9 °C and below (Patel & Crisp, 1960) , its lower critical temperature for feeding activity is 4.6 °C (Southward, 1955). *Semibalanus balanoides* out-competes *Chthamalus* species for space, but recruitment declines and failures of *Semibalanus balanoides* in response to warmer temperatures benefit *Chthamalus* species by allowing them to persist and recruit (Mieszkowska, *et al*., 2014).

The characterizing *Spirobranchus triqueter* are found in both warmer and colder waters experienced in the UK. *Spirobranchus triqueter* occurs from the Arctic, the eastern North Atlantic up to the Mediterranean, Adriatic, Black and Red Sea, the English Channel, the whole North Sea, Skagerrak, Kattegat, the Belts and Öresund up to Bay of Kiel (De Kluijver *et al*., 2016)

The body temperature of *Patella vulgata* can exceed 36°C in the field, (Davies, 1970), adults become become non-responsive at 37-38°C and die at temperatures of 42°C (Evans, 1948). Lower temperatures enhance feeding rates in adults (Thompson *et al*., 2004). Juvenile tolerance of warm air temperatures and dessication may be lower than adults. Juveniles require damp areas of rock (Lewis & Bowmna, 1975) and the bare rock surfaces typical of this biotope, present a harsher habitat than the associated crevices and cracks. Long-term time studies in southern England suggest that *Patella vulgata* have become scarcer following warmer summers, while *Patella depressa* increase in abundance (Southward *et al*., 1995). Increased temperatures may alter spawning cues and reproduction success in *Patella vulgata* populations. Observations suggest that spawning is initiated in autumn storms with greater wave action when seawater temperatures drop below 12°C (Bowman 1985, Bowman & Lewis, 1986, LeQuesne, 2005). In Northern Portugal warming seas appear to be linked to both a shortening of the reproductive period and the lack of multiple spawning events in *Patella vulgata* and other northern species (Ribeiro *et al*., 2009).

The encrusting coralline, *Lithophyllum incrustans*, is close to the northern edge of its reported distribution range in the UK (Kain, 1982; Guiry & Guiry, 2015) and is therefore considered likely to be tolerant of an increase in temperature, particularly in this subtidal biotope, where it is protected from desiccation.

Sensitivity assessment. Typical surface water temperatures around the UK coast vary, seasonally from 4-19°C (Huthnance, 2010). The biotope is considered to tolerate a 2°C increase in temperature for a year. An acute increase at the pressure benchmark may be tolerated in winter, but a sudden return to typical temperatures could lead to mortalities among acclimated animals. No evidence was found to support this assessment, however. An acute increase of 5°C in summer would be close to the lethal thermal temperature for *Balanus crenatus*. Biotope resistance is therefore assessed as ‘Medium’ and resilience as ‘High’ and biotope sensitivity is therefore ‘Low’. Adult *Semibalanus balanoides* are considered likely to be able to tolerate an acute or chronic change, however, if an acute change in temperature occurred in autumn or winter it could disrupt reproduction, while a chronic change could alter reproductive success if it exceeded thermal thresholds for reproduction. The effects would depend on the magnitude, duration and footprint of the activities leading to this pressure. However, barnacle populations are highly connected, with a good larval supply and high dispersal potential (Wethey *et al*., 2011, Rognstad *et al*., 2014). Resistance is therefore assessed as ‘High’ (despite some potential effects on reproductive success) and resilience as ‘High’ (by default). This biotope is therefore considered to be ‘Not sensitive’ at the pressure benchmark, although some changes in the proportions of different barnacle species may occur.

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

Many intertidal species are tolerant of freezing conditions as they are exposed to extremes of low air temperatures during periods of emersion. They must also be able to cope with sharp temperature fluctuations over a short period of time during the tidal cycle. In winter air temperatures are colder than the sea, conversely in summer air temperatures are much warmer than the sea. Species that occur in the intertidal are therefore generally adapted to tolerate a range of temperatures, with the width of the thermal niche positively correlated with the height of the shore that the animal usually occurs at (Davenport & Davenport, 2005).

Spirorbids are distributed circumglobally, with some differences in range between taxonomic groups (Knight-Jones *et al*., 1991). the biotope classification does not specify species or families and therefore it is considered that a spirorbid group may be able to colonize this biotope regardless of temperature changes.

The barnacle *Semibalanus balanoides* is primarily a ‘northern’ species with an arctic-boreal distribution. Long-term time series show that recruitment success is correlated to lower sea temperatures (Mieszkowska *et al*., 2014). Due to warming temperatures its range has been contracting northwards. Temperatures above 10 to 12 °C inhibit reproduction (Barnes, 1957, 1963, Crisp & Patel, 1969) and laboratory studies suggest that temperatures at or below 10 °C for 4-6 weeks are required in winter for reproduction, although the precise threshold temperatures for reproduction are not clear (Rognstad *et al*., 2014). The tolerance of *Semibalanus balanoides* collected in the winter (and thus acclimated to lower temperatures) to low temperatures was tested in the laboratory. The median lower lethal temperature tolerance was -14.6 °C (Davenport & Davenport, 2005). A decrease in temperature at the pressure benchmark is therefore unlikely to negatively affect this species. *Balanus crenatus* is described as a boreal species (Newman & Ross, 1976) it is found throughout the northeast Atlantic from the Arctic to the west coast of France as far south as Bordeaux; east and west coasts of North America and Japan. *Chthamalus stellatus* and *Chthamalus montagui* are ‘southern’ barnacle species and their range has been extending northwards due to increasing temperatures. *Chthamalus* suffers failure of fertilization at temperatures of 9 °C and below (Patel and Crisp, 1960) its lower critical temperature for feeding activity is 4.6 °C (Southward, 1955). The cold winter of 2009-10 in France led to recruitment failure in *Chthamalus* species (Wethey et al., 2011).

The characterizing *Spirobranchus triqueter* are found in both warmer and colder waters experienced in the UK. *Spirobranchus triqueter* occurs from the Arctic, the eastern North Atlantic up to the Mediterranean, Adriatic, Black and Red Sea, the English Channel, the whole North Sea, Skagerrak, Kattegat, the Belts and Öresund up to Bay of Kiel (De Kluijver *et al*., 2016)

The limpet, *Patella vulgata* is largely unaffected by short periods of extreme cold. Ekaratne & Crisp (1984) found adult limpets continuing to grow over winter when temperatures fell to -6°C, and stopped only by still more severe weather. However, loss of adhesion after exposure to -13°C has been observed with limpets falling off rocks and therefore becoming easy prey to crabs or birds (Fretter & Graham, 1994). However, in the very cold winter of 1962-3 when temperatures repeatedly fell below 0°C over a period of 2 months large numbers of *Patella vulgata* were found dead (Crisp, 1964). Periods of frost may also kill juvenile *Patella vulgata*, resulting in recruitment failures in some years (Bowman & Lewis, 1977).

Sensitivity assessment. Based on the wide temperature tolerance range of *the characterizing and* *associated species* it is concluded that the acute and chronic decreases in temperature described by the benchmark would have limited effect on barnacles and limpets. Based on distribution and the link between cooler winter temperatures and reproductive success, *Semibalanus balanoides* is also considered to be unaffected at the pressure benchmark. Changes in recruitment success between southern and northern barnacle species may lead to some changes in the proportions of barnacles species recruiting but this is not considered to alter the character of the biotope. Resistance is therefore assessed as 'High' (despite some potential effects on reproductive success) and resilience as 'High' (by default). This biotope is therefore considered to be 'Not sensitive' at the pressure benchmark

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

This biotope is recorded in full salinity (30-35 ppt) habitats (Connor *et al.*, 2004) and therefore the sensitivity assessment benchmark considers an increase from full salinity to >40 ppt. Biotopes found in the intertidal will naturally experience fluctuations in salinity where evaporation increases salinity and inputs of rainwater expose individuals to freshwater. Species found in the intertidal are therefore likely to have some form of behavioural or physiological adaptations to changes in salinity.

Barnes & Barnes (1974) found that larvae from six barnacle species including *Balanus crenatus*, *Chthamalus stellatus* and *Semibalanus (as Balanus) balanoides*, completed their development to nauplii larvae at salinities between 20-40‰. (Some embryos exposed at later development stages could survive at higher and lower salinities). *Balanus crenatus* occurs in estuarine areas and is therefore adapted to variable salinity (Davenport, 1976). When subjected to sudden changes in salinity *Balanus crenatus* closes its opercular valves so that the blood is maintained temporarily at a constant osmotic concentration (Davenport, 1976).

The crustose corallines that occur in this biotope may also be found on rocky shores and in rockpools where salinities may fluctuate markedly during exposure to the air. Edyvean & Ford (1984b) suggest that populations of *Lithophyllum incrustans* are affected by temperature changes and salinity and that temperature and salinity 'shocks' induce spawning but no information on thresholds was provided (Edyvean & Ford, 1984b). Populations of *Lithophyllum incrustans* were less stable in rockpools with a smaller volume of water that were more exposed to temperature and salinity changes due to lower buffering capacity. Sexual plants (or the spores that give rise to them) were suggested to be more susceptible than asexual plants to extremes of local environmental variables (temperature, salinity, etc.) as they occur with greater frequency at sites where temperature and salinity were more stable (Edyvean & Ford, 1984b).

Local populations may be acclimated to the prevailing salinity regime and may therefore exhibit different tolerances to other populations subject to different salinity conditions and therefore caution should be used when inferring tolerances from populations in different regions. This biotope is found in full (30-35 ppt) salinity (Connor *et al.*, 2004). Biotopes found in the intertidal will naturally experience fluctuations in salinity where evaporation increases salinity and inputs of rainwater expose individuals to freshwater. Species found in the intertidal are therefore likely to have some form of behavioural or physiological adaptations to changes in salinity.

Sensitivity assessment. Little direct evidence was found to assess sensitivity to this pressure. Although some increases in salinity may be tolerated by the associated species present these are generally short-term and mitigated during tidal inundation. This biotope is considered, based on distribution on the mid-shore to be sensitive to a persistent increase in salinity to > 40 ppt. Resistance is therefore assessed as 'Low' and recovery as 'High' (following restoration of usual salinity). Sensitivity is therefore assessed as 'Low'.

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

This biotope is recorded in full salinity (30-35 ppt) (Connor *et al*., 2004). At the pressure benchmark, a change from full to variable salinity (18-30 ppt) is assessed. The characterizing species *Balanus crenatus and Spirobranchus triqueter* are found in a similar biotope (CR.MCR.EcCr.UrtScr), is present in variable salinities (Connor *et al*., 2004). It is therefore likely that the characterizing species will tolerate a reduction in salinity form full to reduced. No direct evidence was found to assess the sensitivity of spirorbids, however, these are described as frequent in the variable salinity biotope LR.LLR.FVS.FserVSIR. and the low salinity biotope LIR.Lag.ProtFur. It is therefore inferred that this group would not be impacted by a reduction in salinity at the pressure benchmark.

Edyvean & Ford (1984b) suggest that populations of the crustose coralline *Lithophyllum incrustans* are affected by temperature changes and salinity and that temperature and salinity 'shocks' induce spawning but no information on thresholds were provided (Edyvean & Ford, 1984b). Populations of *Lithophyllum incrustans* were less stable in tide pools with a smaller volume of water that were more exposed to temperature and salinity changes due to lower buffering capacity. Sexual plants (or the spores that give rise to them) were suggested to be more susceptible than asexual plants to extremes of local environmental variables (temperature, salinity etc.) as they occur with greater frequency at sites where temperature and salinity were more stable (Edyvean & Ford, 1984b).

Balanus crenatus occurs in estuarine areas and is therefore adapted to variable salinity (Davenport, 1976). When subjected to sudden changes in salinity *Balanus crenatus* closes its opercular valves so that the blood is maintained temporarily at a constant osmotic concentration (Davenport, 1976). Acclimation to different salinity regimes alters the point at which opercular closure and resumption of activity occurs (Davenport, 1976). *Balanus crenatus* can tolerate salinities down to 14 psu if given time to acclimate (Foster, 1970). At salinities below 6 psu motor activity ceases, respiration falls and the animal falls in to a "salt sleep". In this state the animals may survive (Barnes & Barnes, 1974) in freshwater for 3 weeks, enabling them to withstand changes in salinity over moderately long periods (Barnes & Powell, 1953). Larvae are more sensitive than adults. In culture experiments, eggs maintained below 10‰ rupture, due to osmotic stress (Barnes & Barnes, 1974). At 15-17‰ there is either no development of early stages or the nauplii larvae are deformed and "probably not viable", similarly at 20‰ development occurs, but about half of the larvae are deformed and not viable. (Barnes & Barnes, 1974). Normal development resulting in viable larvae occurs between salinities of 25-40 ‰ (Barnes & Barnes, 1974). Barnes & Barnes (1965) found that in high suspended solids and low salinity there was a decrease in the number of eggs per brood of *Chthamalus stellatus / Chthamalus montagui* If salinities decrease below 21 psu all cirral activity of barnacles normally associated with full salinity waters, ceases (Foster, 1971).*Semibalanus balanoides* are tolerant of a wide range of salinity and can survive periodic emersion in freshwater, e.g. from rainfall or freshwater run-off, by closing their opercular valves (Foster, 1971b). They can also withstand large changes in salinity over moderately long periods of time by falling into a "salt sleep" and can be found on shores (example from Sweden) with large fluctuations in salinity around a mean of 24 (Jenkins *et al*., 2001).

Spirobranchus triqueter has not been recorded from brackish or estuarine waters. Therefore, it is likely that the species will be very intolerant of a decrease in salinity. However, Dixon (1985, cited in Riley & Ballerstedt, 2005) views the species as able to withstand significant reductions in salinity. The degree of reduction in salinity and time that the species could tolerate those levels were not recorded. Therefore, there is insufficient information available to assess the intolerance of *Spirobranchus triqueter* to a reduction in salinity and the assessment is based on its presence in the biotope CR.MCR.EcCr.UrtScr which occurs in variable salinity (as well as full) habitats (Connor *et al*., 2004).

Sensitivity assessment. As the characterizing species are found in biotopes in both full and variable salinity habitats, the biotope is considered 'Not sensitive' to a decrease in salinity from full to variable. Biotope resistance is therefore assessed as 'High' and resilience is assessed as 'High' (by default) and the biotope is assessed as 'Not sensitive'.

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

The spirorbids, barnacles and coralline crusts characterizing this biotope are securely attached and as these are relatively flat and small they are subject to little or no drag compared to upright growth forms. Colonies of *Lithophyllum incrustans* appear to thrive in conditions exposed to strong water movement (Irvine & Chamberlain, 1994

Changes in flow rate may impact the supply of food to filter feeders. Laboratory experiments demonstrate that barnacle feeding behaviour alters over different flow rates but that barnacles can feed at a variety of flow speeds (Sanford *et al*., 1994). Flow tank experiments using velocities of 0.03, 0.07 and 0.2 m/s showed that a higher proportion of barnacles fed at higher flow rates (Sanford *et al*., 1994). Feeding was passive, meaning the cirri were held out to the flow to catch particles; although active beating of the cirri to gernerate feeding currents occurs in still water (Crisp & Southward, 1961). Field observations at sites in southern New England (USA) that experience a number of different measured flow speeds, found that *Semibalanus balanoides* from all sites responded quickly to higher flow speeds, with a higher proprtion of individuals feeding when current speeds were higher. Barnacles were present at a range of sites, varying from sheltered sites with lower flow rates (maximum observed flow rates <0.06- 0.1 m/s), a bay site with higher flow rates (maximum observed flows 0.2-0.3 m/s) and open coast sites (maximum observed flows 0.2-0.4 m/s). Recruitment was higher at the site with flow rates of 0.2-0.3 m/s (although this may be infuenced by supply) and at higher flow microhabitats within all sites. Both laboratory and field observations indicate that flow is an important factor with effects on feeding, growth and recruitment in *Semibalanus balanoides* (Sanford *et al*., 1994, Leonard *et al*., 1998), however, the results suggest that flow is not a limiting factor determining the overall distribution of barnacles as they can adapt to a variety of flow speeds.

Spirobranchus triqueter is found in biotopes exposed to flow speeds varying from very weak to moderately strong (negligible - >1.5m/s) and was considered 'Not sensitive' at the pressure benchmark (Tillin & Tyler-Walters, 2014). *Balanus crenatus* is found in a very wide range of water flows (Tillin & Tyler-Walters, 2014), although it usually occurs in sites sheltered from wave action (Eckman & Duggins, 1993) and can adapt feeding behaviour according to flow rates. In the absence of any current, the barnacle rhythmically beats its cirri to create a current to collect zooplankton. Growth of *Balanus crenatus* (measured as increase in basal area), maintained for 69 days at constant flow speeds in laboratory experiments was greatest at intermediate flow speeds

(0.08 m/s) and decreased at higher speeds (Eckman & Duggins, 1993). Over the entire range of flow speeds measured (0.02 m/s – 0.25 m/s), *Balanus crenatus*, was able to control the cirrus with little or no deformation by flow observed (Eckman, & Duggins, 1993).

Sensitivity assessment. The species that characterize or are associated with this biotope are securely attached and can occur in a range of flow speeds. Resistance of the biotope to changes in water flow is assessed as ‘High’ and resilience as ‘High’ (by default) so that the biotope is assessed as ‘Not sensitive’. Scour is a key factor structuring this biotope (Connor *et al*., 2004), changes in flow exceeding the pressure benchmark may increase or decrease sediment transport and associated scour may lead to indirect changes in the character of the biotope.

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

Emergence regime is a key factor structuring this (and other) intertidal biotopes. Increased emergence may reduce habitat suitability for characterizing species through greater exposure to dessication and reduced feeding opportunities for the barnacles and other filter feeders including spirobids, barnacles sponges and anemones which feed when immersed. *Semibalanus balanoides* is less tolerant of dessication stress than *Chthamalus* barnacles species and changes in emergence may therefore lead to species replacement and the development of a *Chthamalus* sp. dominated biotope, more typical of the upper shore may develop. Records suggest that, typically, above this biotope the community may grade into a red algal dominated community (AudCla), where light allows (Connor *et al*., 2004). Changes in emergence may therefore eventually lead to replacement of this biotope to one more typical of the upper cave walls.

Decreased emergence would reduce dessication stress and allow the attached suspension feeders more feeding time. Predation pressure on barnacles and limpets is likely to increase where these are submerged for longer periods and to prevent colonisation of lower zones. *Semibalanus balanoides* was able to extend its range into lower zones when protected from predation by the dogwhelk, *Nucella lapillus* (Connell, 1961). The mobile species present within the biotope, including *Patella vulgata*, isopods and *Littorina spp*. would be able to relocate to preferred shore levels. Where decreased emergence leads to increased abrasion and scour while immersed, the removal of fauna may lead to this biotope reverting to the more barren LR.FLR.CvOv.BarCv.

Sensitivity assessment. As emergence is a key factor structuring the distribution of animals on the shore, resistance to a change in emergence (increase or decrease) is assessed as ‘Low’. Recovery is assessed as ‘High’, and sensitivity is therefore assessed as 'Low'.

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

This biotope is recorded from locations that are judged to range from very exposed to sheltered (JNCC, 2015).

The barnacles, spirorbids and crustose corallines associated with this biotope have a flat growth form and are unlikely to be dislodged by increased wave action.

Sensitivity assessment. The biotope and characterizing and associated species are found across a range of wave exposures, populations occurring within the middle of the range are considered to have 'High' resistance to a change in significant wave height at the pressure benchmark. Resilience is assessed as ‘High’, by default, and the biotope is considered ‘Not sensitive’.

Chemical Pressures

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

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

No information was found concerning the effects of heavy metals on encrusting coralline algae. Bryan (1984) suggested that the general order for heavy metal toxicity in seaweeds is: organic Hg> inorganic Hg > Cu > Ag > Zn> Cd> Pb. Contamination at levels greater than the pressure benchmark may adversely impact the biotope. Cole *et al.* (1999) reported that Hg was very toxic to macrophytes. The sub-lethal effects of Hg (organic and inorganic) on the sporelings of an intertidal red algae, *Plumaria elegans*, were reported by Boney (1971). 100% growth inhibition was caused by 1 ppm Hg.

Contamination at levels greater than the pressure benchmark may adversely impact the biotope. Barnacles accumulate heavy metals and store them as insoluble granules (Rainbow, 1987). Pyefinch & Mott (1948) recorded a median lethal concentration of 0.19 mg/l copper and 1.35 mg/l mercury, for *Balanus crenatus* over 24 hours. Barnacles may tolerate fairly high level of heavy metals in nature, for example they are found in Dulas Bay, Anglesey; where copper reaches concentrations of 24.5 µg/l, due to acid mine waste (Foster *et al.*, 1978).

Bryan (1984) suggested that gastropods are also rather tolerant of heavy metals. In the Fal estuary *Patella vulgata* occurs at, or just outside, Restronguet Point at the end of the creek where metal concentrations are in the order: Zinc (Zn) 100-2000µg/l, copper (Cu) 10-100µg/l and cadmium (Cd) 0.25-5µg/l (Bryan & Gibbs, 1983). However, in the laboratory *Patella vulgata* was found to be intolerant of small changes in environmental concentrations of Cd and Zn by Davies (1992). At concentrations of 10µg/l pedal mucus production and levels of activity were both reduced, indicating a physiological response to metal concentrations. Exposure to Cu at a concentration of 100µg/l for one week resulted in progressive brachycardia (slowing of the heart beat) and the death of limpets. Zn at a concentration of 5500µg/l produced the same effect (Marchan *et al.*,1999).

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

Where exposed to direct contact with fresh hydrocarbons, encrusting coralline algae appears to be highly intolerant. Crump *et al.* (1999) described 'dramatic and extensive bleaching' of '*Lithothamnia*' following the *Sea Empress* oil spill. Observations following the *Don Marika* oil spill (K. Hiscock, pers. comm.) were of rockpools with completely bleached coralline algae. However, Chamberlain (1996) observed that although *Lithophyllum incrustans* was affected in a short period of time by oil during the *Sea Empress* spill, recovery occurred within about a year. The oil was found to have destroyed about one third of the thallus thickness but regeneration occurred from thallus filaments below the damaged area.

No information is available on the intolerance of *Balanus crenatus* to hydrocarbons. However, other littoral barnacles generally have a high tolerance to oil (Holt *et al.*, 1995) and were little impacted by the *Torrey Canyon* oil spill (Smith, 1968), so *Balanus crenatus* is probably fairly resistant to oil.

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

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

Barnacles have a low resilience to chemicals such as dispersants, dependant on the concentration and type of chemical involved (Holt *et al.*, 1995). They are less intolerant than some species (e.g. *Patella vulgata*) to dispersants (Southward & Southward, 1978) and *Balanus crenatus* was the dominant species on pier pilings at a site subject to urban sewage pollution (Jakola & Gulliksen, 1987). Hoare & Hiscock (1974) found that *Balanus crenatus* survived near to an acidified halogenated effluent discharge where many other species were killed, suggesting a high tolerance to chemical contamination. Little information is available on the impact of endocrine disrupters on adult barnacles. Holt *et al.* (1995) concluded that barnacles are fairly sensitive to chemical pollution, therefore intolerance is reported as high. The species is an important early colonizer of sublittoral rock surfaces (Kitching, 1937) and it heavily recolonized a site that was dredged for gravel within 7 months (Kenny & Rees, 1994). Therefore, recovery is predicted to be high.

Hoare & Hiscock (1974) reported that the limpet *Patella vulgata* was excluded from sites within 100-150m of the discharge of acidified, halogenated effluent in Amlwch Bay. Limpets are also extremely intolerant of aromatic solvent based dispersants used in oil spill clean-up. During the clean-up response to the *Torrey Canyon* oil spill nearly all the limpets were killed in areas close to dispersant spraying. Viscous oil will not be readily drawn in under the edge of the shell by ciliary currents in the mantle cavity, whereas detergent, alone or diluted in seawater, would creep in much more readily and be liable to kill the limpet (Smith, 1968). A concentration of 5ppm killed half the limpets tested in 24 hours (Southward & Southward, 1978; Hawkins & Southward, 1992). Acidified seawater affects the motility of *Patella vulgata*. At a pH of 5.5 motility was reduced whilst submerged but individuals recovered when returned to normal seawater. At a pH of 2.5 total inhibition of movement occurred and when returned to normal seawater half had died (Bonner *et al.*, 1993). Reduced motility reduces time for foraging and may result in decreased survival of individuals. Acidified seawater can also change the shell composition which will lead to a decrease in its protective nature and hence survival (Bonner *et al.*, 1993). Short periods (48 hours) are unlikely to have much effect on a population but long periods (1 year) may cause reduced grazing and an increase in algal growth. However, seawater is unlikely to reach pH 2.5 therefore intolerance to slight changes in pH will be low. Hoare & Hiscock (1974) reported that in Amlwch Bay *Patella vulgata* was excluded from sites within 100-150m of the discharge of acidified, halogenated effluent.

Most pesticides and herbicides were suggested to be very toxic for invertebrates, especially crustaceans (amphipods isopods, mysids, shrimp and crabs) and fish (Cole *et al.*, 1999).

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

No evidence.

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

Specific information concerning oxygen consumption and reduced oxygen tolerances were not found for the key characterizing species within the biotope. It is likely that as this biotope occurs in areas that are shallow and tidally flushed that re-oxygenation is likely, limiting the effects of any de-oxygenation events. However, this may mean that the species present have little exposure to low oxygen and may be sensitive to this pressure. *Balanus crenatus*, however, respires anaerobically so it can withstand some decrease in oxygen levels. When placed in wet nitrogen, where oxygen stress is maximal and desiccation stress is minimal, *Balanus crenatus* has a mean survival time of 3.2 days (Barnes *et al*., 1963) and this species is considered to be 'Not sensitive' to this pressure. *Semibalanus balanoides* can respire anaerobically, so they can tolerate some reduction in oxygen concentration (Newell, 1979). When placed in wet nitrogen, where oxygen stress is maximal and desiccation stress is low, *Semibalanus balanoides* have a mean survival time of 5 days (Barnes *et al*., 1963).

In laboratory experiments a reduction in the oxygen tension of seawater from 148mm Hg (air saturated seawater) to 50mm Hg rapidly resulted in reduced heart rate in limpets of the genus *Patella* (Marshall & McQuaid, 1993). Heartbeat rate returned to normal in oxygenated water within two hours. Limpets can survive for a short time in anoxic seawater; Grenon & Walker, (1981) found that in oxygen free water limpets could survive up to 36 hours, although Marshall & McQuaid (1989) found a lower tolerance for *Patella granularis*, which survived up to 11 hours in anoxic water. Therefore, some individuals may survive for one week at an oxygen concentration of 2mg/l. However, *Patella vulgata* is able to respire in air, so would only be exposed to low oxygen in the water column intermittently during periods of tidal immersion. In addition, in areas of wave exposure and moderately strong current flow low oxygen levels in the water are unlikely to persist for very long.

Sensitivity assessment. Based on *Balanus crenatus* and *Semibalanus balanoides* and mitigation of de-oxygenation by water movements, this biotope is considered to have 'High' resistance and High resilience (by default), and is therefore 'Not sensitive'.

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

Nutrient enrichment at the pressure benchmark is unlikely to affect the fauna within this biotope. No direct evidence was found to assess this pressure. A slight increase in nutrient levels could be beneficial for barnacles by promoting the growth of phytoplankton levels and therefore increasing zooplankton levels. *Balanus crenatus* was the dominant species on pier pilings, which were subject to urban pollution (Jakola & Gulliksen, 1987). Limpets and other grazers would also benefit from increased growth of benthic microalgae. However, Holt *et al*. (1995) predict that smothering of barnacles by ephemeral green algae is a possibility under eutrophic conditions.

Over geological timescales, periods of increased nutrient availability have experienced increases in the distribution of crustose coralline species at the expense of corals (Littler & Littler, 2013), suggesting that this group have some tolerance for enhanced nutrient levels. Overall, Littler & Littler (2013) suggest that corallines as a group can tolerate both low and elevated levels of nutrients. The encrusting coralline *Lithophyllum incrustans* were present at sites dominated by *Ulva* spp. in the Mediterranean exposed to high levels of nutrient enrichment from domestic sewage (Arévalo *et al*., 2007).

Sensitivity assessment. The pressure benchmark is set at a level that is relatively protective and based on the evidence and considerations outlined above the biological assemblage is considered to be 'Not sensitive' at the pressure benchmark. Resistance and resilience are therefore assessed as 'High'.

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

As the biotope occurs in tide swept or wave exposed areas (Connor *et al*., 2004), water movements will disperse organic matter reducing the level of exposure.

The animals found within the biotope may be able to utilise the input of organic matter as food, or are likely to be tolerant of inputs at the benchmark level. Cabral-Oliveira *et al*., (2014), found that filter feeders including the barnacle *Chthamalus montagui*, were more abundant at sites closer to a sewage treatment works, as they could utilise the organic matter inputs as food. On the same shores, higher abundances of juvenile *Patella* sp. and lower abundances of adults were found closer to sewage inputs, Cabral-Oliveira *et al*., (2014) suggested the structure of these populations was due to increased competition closer to the sewage outfalls.

In a recent review, assigning species to ecological groups based on tolerances to organic pollution, characterizing animal species*; Balanus crenatus and Spirobranchus triqueter* were assigned to AMBI Group II described as 'species indifferent to enrichment, always present in low densities with non-significant variations with time, from initial state, to slight unbalance' (Gittenberger & Van Loon, 2011).

The crusting coralline *Lithophyllum incrustans* were present at sites dominated by *Ulva* spp. in the Mediterranean exposed to high levels of organic pollution from domestic sewage (Arévalo *et al*., 2007), suggesting the encrusting corallines are not sensitive to this pressure.

Sensitivity assessment. It is not clear whether the pressure benchmark would lead to enrichment effects in this dynamic, scoured habitat. High water movements would disperse organic matter particles, mitigating the effect of this pressure. Based on the AMBI categorisation (Borja *et al*., 2000, Gittenberger & Van Loon, 2011), characterizing and associated species are assessed as 'Not Sensitive' to this pressure based on 'High' resistance and 'High' resilience as there is no impact to recover from. Although species within the biotope may be sensitive to gross organic pollution resulting from sewage disposal and aquaculture they are considered to have 'High' resistance to the pressure benchmark (which represents organic enrichment) and therefore 'High' resilience. The biotope is therefore considered to be 'Not Sensitive'.

Physical Pressures

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

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

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

This biotope is characterized by the hard rock substratum to which the key characterizing species, spirobids, barnacles and associated species can firmly attach to. A change to a sedimentary substratum would significantly alter the character of the biotope. More subtle changes in substratum type can also lead to indirect effects. Surface roughness, for example, is correlated with settlement in barnacles (Coombes *et al*., 2015). Spirorbids are also selective and will discriminate between different types of hard surface (James & Underwood, 1994). An artificial substratum may therefore not be equivalent to a natural rock reef habitat. An increase in mobile surfaces can also indirectly decrease suitable habitats. Shanks & Wright (1986) observed that limpet mortalities were much higher at sites where the supply of loose cobbles and pebbles were greater, leading to increased abrasion through wave action 'throwing' rocks onto surfaces. The biotope is therefore considered to have 'No' resistance to this pressure (based on a change to sediments), recovery is assessed as 'Very low', as the change at the pressure benchmark is permanenet. Biotope sensitivity is therefore assessed as 'High'. As this biotope is found in caves, a change in topography from a cave to an open rock surface would also result in the loss of the biotope.

Physical change (to another sediment type)	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR

Not relevant to biotopes occurring on bedrock.

Habitat structure changes - removal of substratum (extraction)	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR

The species characterizing this biotope are epifauna or epiflora occurring on rock and would be sensitive to the removal of the habitat. However, extraction of rock substratum is considered unlikely and this pressure is considered to be ‘Not relevant’ to hard substratum habitats.

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

The species characterizing this biotope occur on the rock and therefore have no protection from surface abrasion. High levels of abrasion from scouring by mobile sands and gravels is an important structuring factor in this biotope (Connor *et al*., 2004) and prevents replacement by less scour-tolerant species, such as shade-tolerant red algae.

The effects of trampling (a source of abrasion) on barnacles appears to be variable with some studies not detecting significant differences between trampled and controlled areas (Tyler-Walters & Arnold, 2008). However, this variability may be related to differences in trampling intensities and abundance of populations studied. The worst case incidence was reported by Brosnan and Crumrine (1994) who reported that a trampling pressure of 250 steps in a 20x20 cm plot one day a month for a period of a year significantly reduced barnacle cover at two study sites. Barnacle cover reduced from 66% to 7% cover in 4 months at one site and from 21% to 5% within 6 months at the second site. Overall barnacles were crushed and removed by trampling. Barnacle cover remained low until recruitment the following spring. Long *et al*. (2011) also found that heavy trampling (70 humans km^{-1} shoreline h^{-1}) led to reductions in barnacle cover. Single step experiments provide a clearer, quantitative indication of sensitivity to direct abrasion. Povey & Keough (1991) in experiments on shores in Mornington peninsula, Victora, Australia, found that in single step experiments 10 out of 67 barnacles, (*Chthamlus antennatus* about 3mm long), were crushed. However, on the same shore, the authors found that limpets may be relatively more resistant to abrasion from trampling. Following step and kicking experiments, few individuals of the limpet C*ellana trasomerica*, (similar size to *Patella vulgata*) suffered damage or relocated (Povey & Keough, 1991). One kicked limpet (out of 80) was broken and 2 (out of 80) limpets that were stepped on could not be relocated the following day (Povey & Keough, 1991). Trampling may lead to indirect effects on limpet populations, Bertocci *et al*., (2011) found that the effects of trampling on *Patella* sp. increased temporal and spatial variability of in abundance. The experimental plots were sited on a wave-sheltered shore dominated by *Ascophyllum nodosum*. On these types of shore, trampling in small patches, that removes macroalgae and turfs, will indirectly enhance habitat suitability for limpets by creating patches of exposed rock for grazing.

Mechanical abrasion from scuba divers was reported to impact encrusting corallines, with cover of *Lithophyllum stictaeforme* greater in areas where diving was forbidden than visited areas (abundance, 6.36 vs 1.4; it is presumed this refers to proportion of cover, although this is not clear from the text, Guarinieri *et al*., 2012). Dethier (1994) experimentally manipulated surface abrasion on a range of encrusting algae including *Lithophyllum impressum*. Crusts were brushed with either a nylon or steel brush for 1 minute each month for 24 months. Unbrushed controls grew by approximately 50% where the cover of nylon brushed crusts and steel brushed crusts decreased by approximately 25% and 40% respectively (interpreted from figures in Dethier, 1994). In laboratory tests on chips of *Lithophyllum impressum* brushing with a steel brush for 1 minute once a week for 3 weeks, resulted in no cover loss of two samples while a third ‘thinned and declined’ (Dethier, 1994).

Hiscock (1983) noted that a community, under conditions of scour and abrasion from stones and boulders moved by storms, developed into a community similar to this biotope, consisting of fast growing species such as *Spirobranchus* (formerly *Pomatoceros*) *triqueter*. Off Chesil Bank, the epifaunal community dominated by *Spirobranchus* (as *Pomatoceros) triqueter* and *Balanus crenatus* decreased in cover in October as it was scoured away in winter storms, but recolonised in May to June (Gorzula 1977). Warner (1985) reported that the community did not contain any persistent individuals but that recruitment was sufficiently predictable to result in a dynamic stability and a similar community, dominated by *Spirobranchus* (as *Pomatoceros) triqueter*, *Balanus crenatus* and *Electra pilosa*, (an encrusting bryozoan), was present in 1979, 1980 and 1983 (Riley and Ballerstedt, 2005).

Shanks & Wright (1986), found that even small pebbles (<6 cm) that were thrown by wave action in Southern California shores could create patches in *Chthamalus fissus* aggregations and could smash owl limpets (*Lottia gigantea*). Average, estimated survivorship of limpets at a wave exposed site, with many loose cobbles and pebbles allowing greater levels of abrasion was 40% lower than at a sheltered site. Severe storms were observed to lead to almost total destruction of local populations of limpets through abrasion by large rocks and boulders.

Sensitivity assessment. The impact of surface abrasion will depend on the footprint, duration and magnitude of the pressure. High levels of abrasion from scouring by mobile cobbles and pebbles is an important structuring factor in this biotope (Connor *et al*., 2004) but the persistence of the assemblage may depend on rapid recovery rather than high resistance (Gorzula, 1977). The evidence for the effects of trampling on barnacles and severe scour on *Balanus crenatus and Spirobranchus triqueter*, suggest that resistance, to a single abrasion event is 'Low' and recovery is 'High'. Sensitivity is assessed as 'Low', based upon the information for these species and the Coralline spp.

Penetration or disturbance of the substratum subsurface	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR

The species characterizing this biotope group are epifauna or epiflora occurring on rock which is resistant to subsurface penetration. The assessment for abrasion at the surface only is therefore considered to equally represent sensitivity to this pressure.

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

This biotope occurs in scoured habitats and it is likely, depending on local sediment supply, that the biotope is exposed to chronic or intermittent episodes of high-levels of suspended solids as local sediments are re-mobilised and transported. A significant increase in suspended solids may result in smothering (see siltation pressures) where these are deposited. Based on Cole *et al*. (1999) and Devlin *et al*. (2008) this biotope is considered to experience intermediate turbidity (10-100 mg/l) based on UK TAG (2014). An increase at the pressure benchmark refers to a change to medium turbidity (100-300 mg/l) and a decrease is assessed as a change to clear (<10 mg/l) based on UK TAG (2014).

An increase in turbidity could be beneficial if the suspended particles are composed of organic matter, however high levels of suspended solids with increased inorganic particles may reduce filter feeding efficiencies. A reduction in suspended solids will reduce food availability for filter feeding species in the biotope (where the solids are organic), although effects are not likely to be lethal over the course of a year. A reduction in light penetration could also reduce growth rate of phytoplankton and so limit zooplankton levels. However, light penetration itself is unlikely to be an important factor as both *Balanus crenatus* and *Spirobranchus triqueter* are recorded from the lower eulittoral or the lower circalittoral.

Available evidence indicates that *Spirobranchus triqueter* is tolerant of a wide range of suspended sediment concentrations (Riley and Ballerstedt, 2005). Stubbings and Houghton (1964) recorded *Spirobranchus* (as *Pomatoceros*) *triqueter* in Chichester harbour, which is a muddy environment. However, *Spirobranchus* (as *Pomatoceros*) *triqueter* has been noted to also occur in areas where there is little or no silt present (Price *et al*., 1980).

Barnes and Bagenal (1951) found that growth rate of *Balanus crenatus* epizoic on *Nephrops norvegicus* was considerably slower than animals on raft exposed panels. This was attributed to reduced currents and increased silt loading of water in the immediate vicinity of *Nephrops norvegicus*. In dredge disposal areas in the Weser estuary, Germany, where turbidity is 35% above the natural rate of 10-100 mg/l, the abundance of *Balanus crenatus* was lower than in reference areas (Witt *et al*., 2004). Separating the effect of increased suspended solids from increased sedimentation and changes in sediment from sediment dumping is problematic, however (Witt *et al*., 2004). Balanids may stop filtration after silt layers of a few millimetres have been discharged (Witt *et al*., 2004), as the feeding apparatus is very close to the sediment surface.

A significant decrease in suspended organic particles may reduce food input to the biotope resulting in reduced growth and fecundity of barnacles and Spirobranchus triqueter. However, local primary productivity may be enhanced where suspended sediments decrease, increasing food supply. Decreased suspended sediment may increase macroalgal competition enhancing diversity but is considered unlikely to significantly change the character of the biotope as colonisation by larger brown macroalgae is limited by the friability of the surface which is unsuitable for attachment.

Gyory *et al*., (2013) found that increased turbidity triggered the release of larvae by *Semibalanus balanoides*, a response which may allow larval release to be timed with high levels of phytoplankton and at times where predation on larvae may be lowered due to the concentration of particles. Storm events that stir up sediments are also associated with larval release (Gyory & Pineda, 2011).

Sensitivity assessment. Overall biotope resistance is assessed as ‘High’ to an increase in suspended solids. Resilience is categorised as ‘High’ (by default). The biotope is considered to be ’Not sensitive’ to decreased suspended solids.

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

As small, sessile species attached to the substratum, siltation at the pressure benchmark would bury spirorbids, barnacles and *Spirobranchus triqueter*. Holme and Wilson (1985) described a *Pomatoceros-Balanus* assemblage on ‘hard surfaces subjected to periodic sever scour and ‘deep submergence by sand or gravel’ in the English Channel. They inferred that the *Pomatoceros-Balanus* assemblage was restricted to fast-growing settlers able to establish themselves in short periods of stability during summer months (Holme and Wilson 1985), as all fauna were removed in the winter months. Barnacles may stop filtration after silt layers of a few millimetres have been discharged as the feeding apparatus is very close to the sediment surface (Witt *et al*., 2004). In dredge disposal areas in the Weser estuary, Germany, where the modelled exposure to sedimentation was 10mm for 25 days, with the centre of the disposal ground exposed to 65 mm for several hours before dispersal, *Balanus crenatus* declined in abundance compared to reference areas. (Witt *et al*., 2004). However, separating the effect of sedimentation from increased suspended solids and changes in sediment from sediment dumping was problematic (Witt *et al*., 2004).

In a review of the effects of sedimentation on rocky coast assemblages, Airoldi (2003) outlined the evidence for the sensitivity of encrusting coralline algae to sedimentation. The reported results are contradictory with some authors suggesting that coralline algae are negatively affected by sediments while others report that encrusting corallines are often abundant or even dominant in a variety of sediment impacted habitats (Airoldi, 2003 and references therein). Crustose corallines have been reported to survive under a turf of filamentous algae and sediment for 58 days (the duration of experiment) in the Galapagos (species not identified, Kendrick, 1991). The crustose coralline *Hydrolithon reinboldii*, has also been reported to survive deposition of silty sediments on subtidal reefs off Hawaii (Littler, 1973). In an experimental study, Balata *et al*. (2007) enhanced sedimentation on experimental plots in the Mediterranean (close to Tuscany) by adding 400 g of fine sediment every 45 days on plots of 400 cm^2 for 1 year. Nearby sites with higher and lower levels of sedimentation were assessed as control plots. Some clear trends were observed. Crustose corallines declined at medium and high levels of sedimentation (Balata *et al*., 2007). The experiment relates to chronic low levels of sedimentation rather than a single acute event, as in the pressure benchmark, however the trends observed are considered to have some relevance to the pressure assessment.

Field observations and laboratory experiments have highlighted the sensitivity of limpets to sediment deposition (see also the ‘heavy’ siltation pressure for further information). Airoldi & Hawkins (2007) tested the effects of different grain sizes and deposit thickness in laboratory experiments using *Patella vulgata*. Sediments were added as a ‘fine’ rain to achieve deposit thicknesses of approximately 1mm, 2 mm, and 4 mm in controlled experiments and grazing and mortality observed over 8-12 days. Limpets were more sensitive to sediments with a higher faction of fines (67% silt) than coarse (58% sand). Coarse sediments of thicknesses approximately 1, 2 and 4 mm decreased grazing activity by 35, 45 and 50 % respectively. At 1 and 2 mm thicknesses, fine sediments decreased grazing by 40 and 77 %. The addition of approximately 4 mm of fine sediment completely inhibited grazing. Limpets tried to escape the sediment but lost attachment and died after a few days (Airoldi & Hawkins, 2007).

Observations on exposed and sheltered shores with patches of sediment around Plymouth in the south west of England ,found that *Patella vulgata* abundances were higher where deposits were absent. The limpets were locally absent in plots with 50-65% sediment cover (Airoldi & Hawkins, 2007). Littler *et al*., (1983) found that the another limpet species, *Lottia gigantea* on southern Californian shores was restricted to refuges from sand burial on shores subject to periodic inundation by sands.

Sensitivity assessment. Based on the presence of the characterizing and associated species in biotopes subject to sedimentation and scour (such as CR.MCR.EcCr.UrtScr), biotope resistance to this pressure, at the benchmark, is assessed as 'High', resilience is assessed as 'High' (by default) and the biotope is considered to be 'Not sensitive'. The assessment considers that sediments are rapidly removed from the biotope and that the scour tolerance of the characterizing animal species and encrusting corallines would prevent significant mortalities although some damage and abrasion may occur. However, if the deposit remained in place; i.e. due to the scale of the pressure or where biotopes were sheltered, or only seasonally subject to water movements or where water flows and wave action were reduced e.g. by the presence of tidal barrages, then resistance would be lower and sensitivity would be greater. Even small deposits of sediments are likely to result in local removal of limpets. The level of impact will depend on the magnitude and duration of impact. Resistance to siltation is assessed as ‘Low’ for *Patella vulgata* based primarily on observations and experiments of Airoldi & Hawkins, (2007), who demonstrated negative effects at deposit thicknesses far lower than the pressure benchmark. Small patches subject to a single impact may recover rapidly via adult migration. As this species is not considered to characterize the biotope, the lower sensitivity based on the barnacles and spirorbids is presented in the risk assessment table.

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

This biotope is described as subject to scouring from abrasion by mobile sediments (Connor *et al*., 2004). The characterizing species occur in biotopes subject to sedimentation and scour (such as CR.MCR.EcCr.UrtScr) and are therefore likely to tolerate intermittent episodes of fine sediment movement and deposition. At the pressure benchmark 'heavy deposition' represents a considerable thickness of deposit and complete burial of the characterizing species would occur. Removal of the sediments by wave action and tidal currents would result in considerable scour. The effect of this pressure will be mediated by the length of exposure to the deposit and the nature of the deposit.

As small, sessile species attached to the substratum, siltation at the pressure benchmark would bury *barnacles and spirorbids*. The lower limits of *Semibalanus balanoides* (as *Balanus balanoides*) appear to be set by levels of sand inundation on sand-affected rocky shores in New Hampshire (Daly & Mathieson, 1977. Holme and Wilson (1985) described a *Pomatoceros-Balanus* assemblage on 'hard surfaces subjected to periodic sever scour and 'deep submergence by sand or gravel' in the English Channel. They inferred that the *Pomatoceros-Balanus* assemblage was restricted to fast-growing settlers able to establish themselves in short periods of stability during summer months (Holme and Wilson, 1985), as all fauna were removed in the winter months. Barnacles may stop filtration after silt layers of a few millimetres have been discharged as the feeding apparatus is very close to the sediment surface (Witt *et al*., 2004). In dredge disposal areas in the Weser estuary, Germany, where the modelled exposure to sedimentation was 10mm for 25 days, with the centre of the disposal ground exposed to 65 mm for several hours before dispersal, *Balanus crenatus* declined in abundance compared to reference areas. (Witt *et al*., 2004). However, separating the effect of sedimentation from increased suspended solids and changes in sediment from sediment dumping was problematic (Witt *et al*., 2004).

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Sensitivity assessment. Sensitivity to this pressure will be mediated by site-specific hydrodynamic conditions and the footprint of the impact. Where a large area is covered sediments may be shifted by wave and tides within the cave rather than removed. Resistance is assessed as 'Medium' as the biotope is exposed to frequent abrasion and scouring (the impact may be mitigated by rapid removal of the deposit) but some removal and mortalities may occur. Resilience is assessed as 'High' based on re-growth from the scour-tolerant, surviving bases of the encrusting corallines and larval recolonization by barnacles, spirorbids and *Spirobranchus triqueter*. Biotope sensitivity is therefore assessed as 'Low'. Even small deposits of sediments are likely to result in local removal of limpets. The level of impact will depend on the magnitude and duration of impact. Resistance to siltation is assessed as 'Low' for *Patella vulgata* based primarily on observations and experiments of Airoldi & Hawkins, (2007), who demonstrated negative effects at deposit thicknesses far lower than the pressure benchmark. Small patches subject to a single impact may recover rapidly via adult migration. As this species is not considered o characterize the biotope, the lower sensitivity based on the barnacles and spirorbids is presented in the risk assessment table.

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

Not assessed. Thompson *et al*., (2004) demonstrated that *Semibalanus balanoides*, kept in aquaria, ingested microplastics within a few days. However, the effects of the microplastics on the health of exposed individuals have not been identified. There is currently no evidence to assess the level of impact. Not relevant. Wave action on exposed shores is likely to generate high levels of underwater noise. Other sources are not considered likely to result in effects on the biotope.

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

No evidence.

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

Not relevant.

Introduction of light or shading	Low Q: High A: Medium C: High	High Q: High A: Low C: High	Low Q: High A: Low C: High

Light penetration is a key factor structuring the cave biotope. Encrusting corallines and other shade-tolerant algae grow closer to the entrance where light availability allows. Encrusting corallines can occur in deeper water than other algae where light penetration is limited. Samples of *Lithophyllum impressum* suspended from a raft and shaded (50-75% light reduction) continued to grow over two years (Dethier, 1994). An increase in light in the spectrum that supports photosynthesis may allow algae including *Rhodochorton purpureum* and *Pilinia maritima* which are found within caves (Connor *et al*., 2004) to colonise more surface area, altering the structure of the biotope. Increased light may also reduce recruitment of spirorbids which have been found to recruit preferentially to shaded areas (Saunders & Connell, 2001).

The animal species are not light dependent. *Spirobranchus triqueter* is found in a variety of light environments from shallow sublittoral biotopes where light levels are relatively high, to deeper sites that are aphotic (De Kluijver, 1993).

Balanus crenatus possesses a rudimentary eye and can detect and respond to sudden shading which may be an anti-predator defence (Forbes *et al*., 1971). *Balanus crenatus* tend to orient themselves when settling, with the least light sensitive area directed towards the light (Forbes *et al*., 1971), so that the more sensitive area can detect shading from predator movements in the area where light availability is lower (Forbes *et al*., 1971).

Semibalanus balanides sheltered from the sun grew bigger than unshaded individuals (Hatton, 1938; cited in Wethey, 1984), although the effect may be due to indirect cooling effects rather than shading. Barnacles are also frquently found under algal canopies suggesting that they are tolerant of shading. Light levels have also been demonstrated to influence a number of phases of the reproductive cycle in *Semibalanus balanoides*. In general light inhibits aspects of the breeding cycle. Penis development is inhibited by light (Barnes & Stone, 1972) while Tighe-Ford (1967) showed that constant light inhibited gonad maturation and fertilization. Davenport & Crisp (unpublished data from Menai Bridge, Wales, cited from Davenport *et al*., 2005) found that experimental exposure to either constant darkness, or 6 h light: 18 h dark photoperiods induced autumn breeding in *Semibalanus*. They also confirmed that very low continuous light intensities (little more than starlight) inhibited breeding. Latitudinal variations in timing of the onset of reproductive phases (egg mass hardening) have been linked to the length of darkness (night) experienced by individuals rather than temperature (Davenport *et al*., 2005). Changes in light levels associated with climate change (increased cloud cover) were considered to have the potential to alter timing of reproduction (Davenport *et al*., 2005) and to shift the range limits of this species southward. However, it is not clear how these findings may reflect changes in light levels from artificial sources, and whether observable changes would occur at the population level as a result. There is, therefore, 'No evidence' on which to base an assessment.

Sensitivity assessment. The key characterizing animal species colonize a broad range of light environments, from intertidal to deeper sub tidal and shaded understorey habitats and are considered to be unaffected by increased shade or more light penetration. However an increase in light in the spectrum that supports photosynthesis may increase algal growth altering the character of the biotope. Some specialist cave species may colonize depending on the presence of source populations. The biotope is therefore considered to have ‘Low’ resistance and ‘High’ resilience following restoration of typical conditions (as crustose corallines and characterizing animals are likely to survive). Sensitivity is therefore considered to be ‘Low’.

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

No direct evidence was found to assess this pressure. Barriers that reduce the degree of tidal excursion may alter larval supply to suitable habitats from source populations. Conversely the presence of barriers may enhance local population supply by preventing the loss of larvae from enclosed habitats. As the larvae of Balanus crenatus and Semibalanus balanoides and other species such as Patella vulgata are planktonic and are transported by water movements, barriers that reduce the degree of tidal excursion may alter larval supply to suitable habitats from source populations. However the presence of barriers may enhance local population supply by preventing the loss of larvae from enclosed habitats. Barriers and changes in tidal excursion are not considered relevant to the characterizing crusting corallines as species dispersal is limited by the rapid rate of settlement and vegetative growth from bases rather than reliance on recruitment from outside of populations. Resistance to this pressure is assessed as 'High' and resilience as 'High' by default. This biotope is therefore considered to be 'Not sensitive'.

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

Not relevant’ to seabed habitats. NB. Collision by grounding vessels is addressed under surface abrasion.

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

Many of the animal species within the biotope probably respond to light levels, detecting shade and shadow to avoid predators, and day length in their behavioural or reproductive strategies. However, their visual acuity is probably very limited and they are unlikely to respond to visual disturbance at the benchmark level. This pressure is, therefore, assessed as ‘Not relevant’.

Balanus crenatus possesses a rudimentary eye and can detect and respond to sudden shading which may be an anti-predator defence (Forbes *et al*., 1971). *Balanus crenatus* tend to orient themselves when settling, with the least light sensitive area directed towards the light (Forbes *et al*., 1971), so that the more sensitive area can detect shading from predator movements in the area where light availability is lower (Forbes *et al*., 1971).

Biological Pressures

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

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

Introduction or spread of invasive non-indigenous species	High Q: High A: High C: High	High Q: High A: High C: High	Not sensitive Q: High A: High C: High

The high levels of scour in this biotope will limit establishment of all but the most scour resistant invasive non-indigenous species (INIS) and no direct evidence was found for effects of INIS on this biotope. The low levels of light within this biotope, particularly the rear walls of caves, are considered to also inhibit invasive algal species.

The Australasian barnacle *Austrominius* (previously *Elminius*) *modestus* was introduced to British waters on ships during the second world war. Increased warming has allowed the Australian barnacle *Austrominius* (formerly, *Elminiusi) modestus*, to dominate sites previously occupied by *Semibalanus balanoides* and *Balanus crenatus* (Witte, 2010). However, on settlement panels deployed in SW Ireland, *Austrominius modestus* initially dominated panels in the lower subtidal but post-recruitment mortality over a year allowed *Balanus crenatus* to become the dominant barnacle (Watson *et al*., 2005). *Balanus crenatus* and *Austrominius modestus* have shown recruitment differences which may alter the seasonal dominance patterns (Witte *et al*., 2010). In generalits overall effect on the dynamics of rocky shores has been small as *Austrominius modestus* has simply replaced some individuals of a group of co-occurring barnacles (Raffaelli & Hawkins, 1999). Although present, monitoring indicates it has not outnumbered native barnacles in the Isle of Cumbrae (Gallagher *et al*., 2015), it may dominate in estuaries where it is more tolerant of lower salinities than *Semibalanus balanoides* (Gomes-Filho, *et al*., 2010).

Two non-native spirobids – *Dexiospira oshoroensis* and *Pileolaria rosepigmentata* - were found on the non-native algae *Sargassum muticum* in Portsmouth (Knight-Jones *et al*., 1975). Invasive tubeworms are reported from UK harbours (Thorp *et al*., 1986) and are likely to be well established in areas with large volumes of ship traffic.

The tunicates *Didemnum vexillum* and *Asterocarpa humilis*, the hydroid *Schizoporella japonica* and the bryozoan *Watersipora subatra* (Bishop, 2012c, Bishop, 2015a and b; Wood, 2015) are currently only recorded from artificial hard substratum in the UK and it is not clear what their established range and impacts in the UK would be. *Didemnum vexillum* occurs in tide pools in other areas where it has become established (Bishop, 2012c) and can have substantial effects on communities, similarly the tunicates *Corella eumyota* and *Botrylloides violaceus* can smother rock habitats (Bishop, 2011b and 2012b).

Sensitivity assessment. Overall, there is little evidence of this biotope being adversely affected by non-native species and scouring of this biotope by mobile sediments is likely to limit establishment of all but robust species. Replacement of native barnacles and spirorbids by non-natives alters the identity of the species present but has little impact on biotope character and function. Resistance is therefore assessed as 'High', and resilience as 'High' (by default), and the biotope is considered to be 'Not sensitive'.

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

No evidence was found that microbial pathogens cause high levels of disease or mortality in this biotope.

The cherizing species *Semibalanus balanoides* are considered subject to persistent, low levels of infection by pathogens and parasites. Barnacles are parasitised by a variety of organisms and, in particular, the cryptoniscid isopod *Hemioniscus balani*, in which heavy infestation can cause castration of the barnacle. At usual levels of infestation these are not considered to lead to high levels of mortality. The associated species *Patella vulgata* has been reported to be infected by the protozoan *Urceolaria patellae* (Brouardel, 1948) at sites sheltered from extreme wave action in Orkney. Baxter (1984) found shells to be infested with two boring organisms, the polychaete *Polydora ciliata* and a siliceous sponge *Cliona celata*.

Diseased encrusting corallines were first observed in the tropics in the early 1990's when the bacterial pathogen Coralline Lethal Orange Disease (CLOD) was discovered (Littler & Littler, 1995). All species of articulated and crustose species tested to date are easily infected by CLOD and it has been increasing in occurrence at sites where first observed and spreading through the tropics. Another bacterial pathogen causing a similar CLOD disease has been observed with a greater distribution and a black fungal pathogen first discovered in American Samoa has been dispersing (Littler & Littler, 1998). An unknown pathogen has also been reported to lead to white 'target-shaped' marks on corallines, again in the tropic (Littler *et al*., 2007). No evidence was found that these pathogens are impacting temperate coralline habitats.

Sensitivity assessment. Based on the lack of reported mortalities of the characterizing species and the available evidence for the coralline crust, the biotope is judged to have 'High' resistance to this pressure. By default resilience is assessed as 'High' and the biotope is classed as 'Not sensitive' at the pressure benchmark.

Removal of target species	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR

Direct, physical impacts from harvesting are assessed through the abrasion and penetration of the seabed pressures. The sensitivity assessment for this pressure considers any biological/ecological effects resulting from the removal of target species on this biotope. Limpets may be gathered recreationally for consumption but removal of this species is not considered to alter the character of the biotope through its loss as shade and scour, rather than grazing, are the key factors limiting the presence of algae (Connor *et al*., 2004). No commercial application or harvesting of other characterizing or associated species is described in the literature and this pressure is therefore considered to be 'Not relevant'.

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

Incidental removal of the key characterizing species would alter the character of the biotope, resulting in reclassification and the loss of species richness. The ecological services such as primary and secondary production, provided by characterizing and associated species, would also be lost. As most species present in this biotope are relatively large, conspicuous and either sedentary or attached to rock surfaces that have little protection against removal.

Sensitivity assessment. Removal of a large percentage of the characterizing species resulting in bare rock would alter the character of the biotope, species richness and ecosystem function. Resistance is therefore assessed as ‘Low’ and recovery as ‘High’, so that biotope sensitivity is assessed as 'Low’.

Bibliography

Adey, W.H. & Adey, P.J., 1973. Studies on the biosystematics and ecology of the epilithic crustose corallinacea of the British Isles. *British Phycological Journal*, **8**, 343-407.

Airoldi, L., 2003. The effects of sedimentation on rocky coast assemblages. *Oceanography and Marine Biology: An Annual Review*, **41**,161-236

Airoldi, L., 2000. Responses of algae with different life histories to temporal and spatial variability of disturbance in subtidal reefs. *Marine Ecology Progress Series*, **195** (8), 81-92.

Airoldi, L. & Hawkins, S.J., 2007. Negative effects of sediment deposition on grazing activity and survival of the limpet *Patella vulgata*. *Marine Ecology Progress Series*, **332**, 235-240.

Andersson, M.H., Berggren, M., Wilhelmsson, D. & Öhman, M.C., 2009. Epibenthic colonization of concrete and steel pilings in a cold-temperate embayment: a field experiment. *Helgoland Marine Research*, **63**, 249-260.

Arévalo, R., Pinedo, S. & Ballesteros, E., 2007. Changes in the composition and structure of Mediterranean rocky-shore communities following a gradient of nutrient enrichment: descriptive study and test of proposed methods to assess water quality regarding macroalgae. *Marine Pollution Bulletin*, **55** (1), 104-113.

Balata, D., Piazzi, L. & Cinelli, F., 2007. Increase of sedimentation in a subtidal system: effects on the structure and diversity of macroalgal assemblages. *Journal of Experimental Marine Biology and Ecology*, **351**(1), 73-82.

Barnes, H., 1956. *Balanus balanoides* (L.) in the Firth of Clyde: the development and annual variation in the larval population and the causative factors. *Journal of Animal Ecology*, **25**, 72-84.

Barnes, H. & Bagenal, T.B., 1951. Observations on *Nephrops norvegicus* and an epizoic population of *Balanus crenatus*. *Journal of the Marine Biological Association of the United Kingdom*, **30**, 369-380.

Barnes, H. & Barnes, M., 1974. The responses during development of the embryos of some common cirripedes to wide changes in salinity. *Journal of Experimental Marine Biology and Ecology*, **15** (2), 197-202.

Barnes, H. & Barnes, M., 1968. Egg numbers, metabolic efficiency and egg production and fecundity; local and regional variations in a number of common cirripedes. *Journal of Experimental Marine Biology and Ecology*, **2**, 135-153.

Barnes, H. & Powell, H.T., 1953. The growth of *Balanus balanoides* and *B. crenatus* under varying conditions of submersion. *Journal of the Marine Biological Association of the United Kingdom*, **32**, 107-127.

Barnes, H. & Stone, R., 1972. Suppression of penis development in *Balanus balanoides* (L.). *Journal of Experimental Marine Biology and Ecology*, **9** (3), 303-309.

Barnes, H., 1957. Processes of restoration and synchronization in marine ecology. The spring diatom increase and the 'spawning' of the common barnacle *Balanus balanoides* (L.). *Année Biologique. Paris*, **33**, 68-85.

Barnes, H., 1963. Light, temperature and the breeding of *Balanus balanoides*. *Journal of the Marine Biological Association of the United Kingdom*, **43** (03), 717-727.

Barnes, H., Finlayson, D.M. & Piatigorsky, J., 1963. The effect of desiccation and anaerobic conditions on the behaviour, survival and general metabolism of three common cirripedes. *Journal of Animal Ecology*, **32**, 233-252.

Baxter, J.M., 1984. The incidence of *Polydora ciliata* and *Cliona celata* boring the shell of *Patella vulgata* in Orkney. *Journal of the Marine Biological Association of the United Kingdom*, **64**, 728-729.

Bennell, S.J., 1981. Some observations on the littoral barnacle populations of North Wales. *Marine Environmental Research*, **5**, 227-240.

Bertness, M.D., Gaines, S.D., Bermudez, D. & Sanford, E., 1991. Extreme spatial variation in the growth and reproductive output of the acorn barnacle *Semibalanus balanoides*. *Marine Ecology Progress Series*, 75, 91-100.

Bertocci, I., Araujo, R., Vaselli, S. & Sousa-Pinto, I., 2011. Marginal populations under pressure: spatial and temporal heterogeneity of *Ascophyllum nodosum* and associated assemblages affected by human trampling in Portugal. Marine Ecology Progress Series, **439**, 73-82.

Bertocci, I., Arenas, F., Matias, M., Vaselli, S., Araújo, R., Abreu, H., Pereira, R., Vieira, R. & Sousa-Pinto, I., 2010. Canopy-forming species mediate the effects of disturbance on macroalgal assemblages on Portuguese rocky shores. *Marine Ecology Progress Series*, **414**, 107-116.

Bishop J. 2011b.Orange-tipped sea squirt, *Corella eumyota*. Great Britain Non-native Species Secretariat. [cited 16/06/2015]. Available from: http://www.nonnativespecies.org

Bishop, J. 2012c. Carpet Sea-squirt, *Didemnum vexillum.Great Britain Non-native Species Secretariat* [On-line]. [cited 30/10/2018]. Available from: http://www.nonnativespecies.org/factsheet/factsheet.cfm?speciesId=1209

Bishop, J. 2012b. *Botrylloides violaceus*. *Great Britain Non-native Species Secretariat*. [On-line] [cited 16/06/2015]. Available from:

Bishop, J. 2015a. Compass sea squirt, *Asterocarpa humilis*. *Great Britain Non-native Species Secretariat*. [On-line] [cited 16/06/2015]. Available from: <http://www.nonnativespecies.org

Bishop, J. 2015b. *Watersipora subatra*. *Great Britain Non-native Species Secretariat*. [On-line][cited 16/06/2015]. Available from: <http://www.nonnativespecies.org

Boney, A.D., 1971. Sub-lethal effects of mercury on marine algae. *Marine Pollution Bulletin*, **2**, 69-71.

Bonner, T. M., Pyatt, F. B. & Storey, D. M., 1993. Studies on the motility of the limpet *Patella vulgata* in acidified sea-water. *International Journal of Environmental Studies*, **43**, 313-320.

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.

Bowman, R.S., 1985. The biology of the limpet *Patella vulgata* L. in the British Isles: spawning time as a factor determining recruitment sucess. In *The Ecology of Rocky Coasts: essays presented to J.R. Lewis, D.Sc.*, (ed. P.G. Moore & R. Seed), Hodder and Stoughton, London, pages 178-193.

Bowman, R.S. and Lewis, J.R., 1986. Geographical variation in the breeding cycles and recruitment of *Patella* spp. *Hydrobiologia*, **142**, 41-56.

Bowman, R.S. & Lewis, J.R., 1977. Annual fluctuations in the recruitment of *Patella vulgata* L. *Journal of the Marine Biological Association of the United Kingdom*, **57**, 793-815.

Bradshaw, C., Veale, L.O., Hill, A.S. & Brand, A.R., 2002. The role of scallop-dredge disturbance in long-term changes in Irish Sea benthic communities: a re-analysis of an historical dataset. *Journal of Sea Research*, **47**, 161-184.

Brault, S. & Bourget, E., 1985. Structural changes in an estuarine subtidal epibenthic community: biotic and physical causes. *Marine Ecology Progress Series*, **21**, 63-73.

Brosnan, D.M. & Crumrine, L.L., 1992. Human impact and a management strategy for Yaquina Head Outstanding Natural Area (summary only). A report to the Bureau of Land Management, Department of the Interior, Salem, Oregon.

Brouardel, J., 1948. Etude du mode d'infestation des Patelles par *Urceolaria patellae* (Cuenot): influence de l'espece de Patelle. *Bulletin du Laboratoire maritime de Dinard*, **30**, 1-6.

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

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

Cabral-Oliveira, J., Mendes, S., Maranhão, P. & Pardal, M., 2014. Effects of sewage pollution on the structure of rocky shore macroinvertebrate assemblages. *Hydrobiologia*, **726** (1), 271-283.

Cadée, G.C., 2007. Balanuliths: free-living clusters of the barnacle *Balanus crenatus*. *Palaios*, **22**, 680-681.

Campbell, D.A. & Kelly, M.S., 2002. Settlement of *Pomatoceros triqueter* (L.) in two Scottish lochs, and factors determining its abundance on mussels grown in suspended culture. *Journal of Shellfish Research*, **21**, 519-528.

Castric-Fey, A., 1983. Recruitment, growth and longevity of *Pomatoceros triqueter* and *Pomatoceros lamarckii* (Polychaeta, Serpulidae) on experimental panels in the Concarneau area, South Brittany. *Annales de l'Institut Oceanographique, Paris*, **59**, 69-91.

Chamberlain, Y.M., 1996. Lithophylloid Corallinaceae (Rhodophycota) of the genera *Lithophyllum* and *Titausderma* from southern Africa. *Phycologia*, **35**, 204-221.

Cole, S., Codling, I.D., Parr, W., Zabel, T., 1999. Guidelines for managing water quality impacts within UK European marine sites [On-line]. *UK Marine SACs Project*. [Cited 26/01/16]. Available from: http://www.ukmarinesac.org.uk/pdfs/water_quality.pdf

Colhart, B.J., & Johanssen, H.W., 1973. Growth rates of *Corallina officinalis* (Rhodophyta) at different temperatures. *Marine Biology*, **18**, 46-49.

Collie, J.S., Hermsen, J.M., Valentine, P.C. & Almeida, F.P., 2005. Effects of fishing on gravel habitats: assessment and recovery of benthic megafauna on Georges Bank. *American Fisheries Society Symposium, American Fisheries Society*, **41**, pp. 325.

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/

Constantino, R., Gaspar, M., Tata-Regala, J., Carvalho, S., Cúrdia, J., Drago, T., Taborda, R. & Monteiro, C., 2009. Clam dredging effects and subsequent recovery of benthic communities at different depth ranges. *Marine Environmental Research*, **67**, 89-99.

Coombes, M.A., La Marca, E.C., Naylor, L.A. & Thompson, R.C., 2015. Getting into the groove: Opportunities to enhance the ecological value of hard coastal infrastructure using fine-scale surface textures. *Ecological Engineering*, **77**, 314-323.

Cotter, E., O'Riordan, R.M & Myers, A.A. 2003. Recruitment patterns of serpulids (Annelida: Polychaeta) in Bantry Bay, Ireland. *Journal of the Marine Biological Association of the United Kingdom*, **83**, 41-48.

Crisp, D. & Patel, B., 1969. Environmental control of the breeding of three boreo-arctic cirripedes. *Marine Biology*, **2** (3), 283-295.

Crisp, D.J. & Southward, A.J., 1961. Different types of cirral activity *Philosophical Transactions of the Royal Society of London, Series B*, **243**, 271-308.

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.

Crisp, D.J., 1964b. Mortalities in marine life in North Wales during the winter of 1962-63. *Journal of Animal Ecology*, **33**, 190-197.

Crisp, D.J., 1965. The ecology of marine fouling. In: *Ecology and the Industrial Society, 5th Symposium of the British Ecological Society, 99-117* (ed. G.T. Goodman, R.W. Edwards & J.M. Lambert).

Crump, R.G., Morley, H.S., & Williams, A.D., 1999. West Angle Bay, a case study. Littoral monitoring of permanent quadrats before and after the *Sea Empress* oil spill. *Field Studies*, **9**, 497-511.

Daly, M.A. & Mathieson, A.C., 1977. The effects of sand movement on intertidal seaweeds and selected invertebrates at Bound Rock, New Hampshire, USA. *Marine Biology*, **43**, 45-55.

Davenport, J., 1976. A comparative study of the behaviour of some balanomorph barnacles exposed to fluctuating sea water concentrations. *Journal of the Marine Biological Association of the United Kingdom*, **5**, pp.889-907.

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

Davenport, J., Berggren, M.S., Brattegard, T., Brattenborg, N., Burrows, M., Jenkins, S., McGrath, D., MacNamara, R., Sneli, J.-A. & Walker, G., 2005. Doses of darkness control latitudinal differences in breeding date in the barnacle *Semibalanus balanoides*. *Journal of the Marine Biological Association of the United Kingdom*, **85** (01), 59-63.

Davies, M.S., 1992. Heavy metals in seawater: effects on limpet pedal mucus production. *Water Research*, **26**, 1691-1693.

Davies, S.P., 1970. Physiological ecology of *Patella* IV. Environmental and limpet body temperatures. *Journal of the Marine Biological Association of the United Kingdom*, **50** (04), 1069-1077.

De Kluijver, M.J., 1993. Sublittoral hard-substratum communities off Orkney and St Abbs (Scotland). *Journal of the Marine Biological Association of the United Kingdom*, **73** (4), 733-754.

Dethier, M.N., 1994. The ecology of intertidal algal crusts: variation within a functional group. *Journal of Experimental Marine Biology and Ecology*, **177** (1), 37-71.

Devlin, M.J., Barry, J., Mills, D.K., Gowen, R.J., Foden, J., Sivyer, D. & Tett, P., 2008. Relationships between suspended particulate material, light attenuation and Secchi depth in UK marine waters. *Estuarine, Coastal and Shelf Science*, **79** (3), 429-439.

Dixon, D.R., 1985. Cytogenetic procedures. *Pomatoceros triqueter*: A test system for environmental mutagenesis. In *The effects of stress and pollution in marine animals*.

Donovan, S.K., 2011. Postmortem encrustation of the alien bivalve *Ensis americanus* (Binney) by the barnacle Balanus crenatus Brugière in the North Sea. *Palaios*, **26**, 665-668.

Dons, C., 1927. Om Vest og voskmåte hos *Pomatoceros triqueter*. *Nyt Magazin for Naturvidenskaberne*, **LXV**, 111-126.

Eckman, J.E. & Duggins, D.O., 1993. Effects of flow speed on growth of benthic suspension feeders. *Biological Bulletin*, **185**, 28-41.

Edyvean, R. & Ford, H., 1986. Spore production by *Lithophyllum incrustans* (Corallinales, Rhodophyta) in the British Isles. *British Phycological Journal*, **21** (3), 255-261.

Edyvean, R.G.J. & Ford, H., 1987. Growth rates of *Lithophyllum incrustans* (Corallinales, Rhodophyta) from south west Wales. *British Phycological Journal*, **22** (2), 139-146.

Edyvean, R.G.J. & Ford, H., 1984a. Population biology of the crustose red alga *Lithophyllum incrustans* Phil. 2. A comparison of populations from three areas of Britain. *Biological Journal of the Linnean Society*, **23** (4), 353-363.

Edyvean, R.G.J. & Ford, H., 1984b. Population biology of the crustose red alga *Lithophyllum incrustans* Phil. 3. The effects of local environmental variables. *Biological Journal of the Linnean Society*, **23**, 365-374.

Ekaratne, S.U.K. & Crisp, D.J., 1984. Seasonal growth studies of intertidal gastropods from shell micro-growth band measurements, including a comparison with alternative methods. *Journal of the Marine Biological Association of the United Kingdom*, **64**, 183-210.

Evans, R.G., 1948. The lethal temperatures of some common British littoral molluscs. *The Journal of Animal Ecology*, **17**, 165-173.

Fernandez-Leborans, G. & Gabilondo, R., 2006. Taxonomy and distribution of the hydrozoan and protozoan epibionts on *Pagurus bernhardus* (Linnaeus, 1758) (Crustacea, Decapoda) from Scotland. *Acta Zoologica*, **87**, 33-48.

Forbes, L., Seward, M.J. & Crisp, D.J., 1971. Orientation to light and the shading response in barnacles. In: *Proceedings of the 4th European Marine Biology Symposium*. Ed. Crisp, D.J., Cambridge University Press, Cambridge. pp 539-558.

Foster, B.A., 1970. Responses and acclimation to salinity in the adults of some balanomorph barnacles. *Philosophical Transactions of the Royal Society of London, Series B*, **256**, 377-400.

Foster, B.A., 1971b. On the determinants of the upper limit of intertidal distribution of barnacles. *Journal of Animal Ecology*, **40**, 33-48.

Foster, P., Hunt, D.T.E. & Morris, A.W., 1978. Metals in an acid mine stream and estuary. *Science of the Total Environment*, **9**, 75-86.

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

Gallagher, M.C., Davenport, J., Gregory, S., McAllen, R. & O'Riordan, R., 2015. The invasive barnacle species, *Austrominius modestus*: Its status and competition with indigenous barnacles on the Isle of Cumbrae, Scotland. *Estuarine, Coastal and Shelf Science*, **152**, 134-141.

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

Gomes-Filho, J., Hawkins, S., Aquino-Souza, R. & Thompson, R., 2010. Distribution of barnacles and dominance of the introduced species *Elminius modestus* along two estuaries in South-West England. *Marine Biodiversity Records*, **3**, e58.

Gorzula, S., 1977. A study of growth in the brittle-star *Ophiocomina nigra*. *Western Naturalist*, **6**, 13-33.

Grenon, J.F. & Walker, G., 1981. The tenacity of the limpet, *Patella vulgata* L.: an experimental approach. *Journal of Experimental Marine Biology and Ecology*, **54**, 277-308.

Guarnieri, G., Terlizzi, A., Bevilacqua, S. & Fraschetti, S., 2012. Increasing heterogeneity of sensitive assemblages as a consequence of human impact in submarine caves. *Marine Biology*, **159** (5), 1155-1164.

Guiry, M.D. & Guiry, G.M. 2015. AlgaeBase [Online], National University of Ireland, Galway [cited 30/6/2015]. Available from: http://www.algaebase.org/

Gyory, J. & Pineda, J., 2011. High-frequency observations of early-stage larval abundance: do storms trigger synchronous larval release in *Semibalanus balanoides*? *Marine Biology*, **158** (7), 1581-1589.

Gyory, J., Pineda, J. & Solow, A., 2013. Turbidity triggers larval release by the intertidal barnacle *Semibalanus balanoides*. *Marine Ecology Progress Series*, **476**, 141-151.

Harms, J. & Anger, K., 1983. Seasonal, annual, and spatial variation in the development of hard bottom communities. *Helgoländer Meeresuntersuchungen*, **36**, 137-150.

Hatcher, A.M., 1998. Epibenthic colonization patterns on slabs of stabilised coal-waste in Poole Bay, UK. *Hydrobiologia*, **367**, 153-162.

Hatton,H., 1938. Easelsde bionomie explicative surquelques especesintercotidalesd'algues et d'animaux. *Annales de l'Institut Oceanographique* **17**: 241-348

Hawkins, S.J. & Southward, A.J., 1992. The *Torrey Canyon* oil spill: recovery of rocky shore communities. In *Restoring the Nations Marine Environment, (ed. G.W. Thorpe), Chapter 13, pp. 583-631*. Maryland, USA: Maryland Sea Grant College.

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

Heath, D., 1976. The distribution and orientation of epizoic barnacles on crabs. *Zoological Journal of the Linnean Society*, **59**, 59-67.

Hiscock, K., 1983. Water movement. In *Sublittoral ecology. The ecology of shallow sublittoral benthos* (ed. R. Earll & D.G. Erwin), pp. 58-96. Oxford: Clarendon Press.

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

Holme, N.A. & Wilson, J.B., 1985. Faunas associated with longitudinal furrows and sand ribbons in a tide-swept area in the English Channel. *Journal of the Marine Biological Association of the United Kingdom*, **65**, 1051-1072.

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.

Hudon, C., Bourget, E., & Legendre, P., 1983. An integrated study of the factors influencing the choice of the settling site of *Balanus crenatus* cyprid larvae. *Canadian Journal of Fisheries and Aquatic Sciences*, **40** (8), 1186-1194.

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

Irvine, L. M. & Chamberlain, Y. M., 1994. *Seaweeds of the British Isles*, vol. 1. *Rhodophyta*, Part 2B *Corallinales, Hildenbrandiales.* London: Her Majesty's Stationery Office.

Jakola, K.J. & Gulliksen, B., 1987. Benthic communities and their physical environment to urban pollution from the city of Tromso, Norway. *Sarsia*, **72**, 173-182.

James, R. & Underwood, A., 1994. Influence of colour of substratum on recruitment of spirorbid tubeworms to different types of intertidal boulders. *Journal of Experimental Marine Biology and Ecology*, **181** (1), 105-115.

Jenkins, S., Åberg, P., Cervin, G., Coleman, R., Delany, J., Della Santina, P., Hawkins, S., LaCroix, E., Myers, A. & Lindegarth, M., 2000. Spatial and temporal variation in settlement and recruitment of the intertidal barnacle *Semibalanus balanoides* (L.)(Crustacea: Cirripedia) over a European scale. *Journal of Experimental Marine Biology and Ecology*, **243** (2), 209-225.

Jenkins, S.R., Beukers-Stewart, B.D. & Brand, A.R., 2001. Impact of scallop dredging on benthic megafauna: a comparison of damage levels in captured and non-captured organisms. *Marine Ecology Progress Series*, **215**, 297-301.

Jenkins, S.R., Hawkins, S.J. & Norton, T.A., 1999. Direct and indirect effects of a macroalgal canopy and limpet grazing in structuring a sheltered inter-tidal community. *Marine Ecology Progress Series*, **188**, 81-92.

Jensen, A.C., Collins, K.J., Lockwood, A.P.M., Mallinson, J.J. & Turnpenny, W.H., 1994. Colonization and fishery potential of a coal-ash artificial reef, Poole Bay, United Kingdom. *Bulletin of Marine Science*, **55**, 1263-1276.

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

Kain, J.M., 1982. The reproductive phenology of nine species of the Rhodophycota in the subtidal region of the Isle of Man. *British Phycological Journal*, **17**, 321-331.

Kain, J.M., 1987. Photoperiod and temperature as triggers in the seasonality of *Delesseria sanguinea*. *Helgolander Meeresuntersuchungen*, **41**, 355-370.

Kaiser, M.J., Cheney, K., Spence, F.E., Edwards, D.B. & Radford, K., 1999. Fishing effects in northeast Atlantic shelf seas: patterns in fishing effort, diversity and community structure VII. The effects of trawling disturbance on the fauna associated with the tubeheads of serpulid worms. *Fisheries Research (Amsterdam)*, **40**, 195-205.

Kaliszewicz, A., Panteleeva, N., Olejniczak, I., Boniecki, P. and Sawicki, M., 2012. Internal brooding affects the spatial structure of intertidal sea anemones in the Arctic-boreal region. *Polar biology*, **35** (12), pp.1911-1919.

Kendall, M.A., Bowman, R.S., Williamson, P. & Lewis, J.R., 1985. Annual variation in the recruitment of *Semibalanus balanoides* on the North Yorkshire coast 1969-1981. *Journal of the Marine Biological Association of the United Kingdom*, **65**, 1009-1030.

Kendrick, G.A., 1991. Recruitment of coralline crusts and filamentous turf algae in the Galapagos archipelago: effect of simulated scour, erosion and accretion. *Journal of Experimental Marine Biology and Ecology*, **147** (1), 47-63

Kenny, A.J. & Rees, H.L., 1994. The effects of marine gravel extraction on the macrobenthos: early post dredging recolonisation. *Marine Pollution Bulletin*, **28**, 442-447.

Kitching, J.A., 1937. Studies in sublittoral ecology. II Recolonization at the upper margin of the sublittoral region; with a note on the denudation of *Laminaria* forest by storms. *Journal of Ecology*, **25**, 482-495.

Knight-Jones, P., Knight-Jones, E. & Buzhinskaya, G., 1991. Distribution and interrelationships of northern spirorbid genera. *Bulletin of Marine Science*, **48** (2), 189-197.

Knight-Jones, P., Knight-Jones, E.W., Thorp, C.H. & Gray, P.W.G., 1975. Immigrant Spirorbids (Polychaeta Sedentaria) on the Japanese *Sargassum* at Portsmouth, England. *Zoologica Scripta*, **4** (1), 145-149.

Leonard, G.H., Levine, J.M., Schmidt, P.R. & Bertness, M.D., 1998. Flow-driven variation in intertidal community structure in a Maine estuary. *Ecology*, **79** (4), 1395-1411.

Le Quesne W.J.F. 2005. *The response of a protandrous species to exploitation, and the implications for management: a case study with patellid limpets.* PhD thesis. University of Southampton, Southampton, United Kingdom.

Lewis, J. & Bowman, R.S., 1975. Local habitat-induced variations in the population dynamics of *Patella vulgata* L. *Journal of Experimental Marine Biology and Ecology*, **17** (2), 165-203.

Littler, M. & Littler, D., 1998. An undescribed fungal pathogen of reef-forming crustose corraline algae discovered in American Samoa. *Coral Reefs*, **17** (2), 144-144.

Littler, M. & Littler, D.S. 2013. The nature of crustose coralline algae and their interactions on reefs. *Smithsonian Contributions to the Marine Sciences*, **39**, 199-212

Littler, M.M., 1973. The population and community structure of Hawaiian fringing-reef crustose Corallinaceae (Rhodophyta, Cryptonemiales). *Journal of Experimental Marine Biology and Ecology*, **11** (2), 103-120.

Littler, M.M. & Littler, D.S., 1995. Impact of CLOD pathogen on Pacific coral reefs. *Science*, **267**, 1356-1356.

Littler, M.M., Littler, D.S. & Brooks, B.L. 2007. Target phenomena on south Pacific reefs: strip harvesting by prudent pathogens? *Reef Encounter*, **34**, 23-24

Littler, M.M., Martz, D.R. & Littler, D.S., 1983. Effects of recurrent sand deposition on rocky intertidal organisms: importance of substrate heterogeneity in a fluctuating environment. *Marine Ecology Progress Series*. **11** (2), 129-139.

Long, J.D., Cochrane, E. & Dolecal, R., 2011. Previous disturbance enhances the negative effects of trampling on barnacles. *Marine Ecology Progress Series*, **437**, 165-173.

Luther, G., 1987. Seepocken der deutschen Küstengewässer. *Helgol Meeresunters* **41**, 1–43

Marchan, S., Davies, M.S., Fleming, S. & Jones, H.D., 1999. Effects of copper and zinc on the heart rate of the limpet *Patella vulgata* (L.) *Comparative Biochemistry and Physiology*, **123A**, 89-93.

Marshall, D.J. & McQuaid, C.D., 1989. The influence of respiratory responses on the tolerance to sand inundation of the limpets *Patella granularis* L.(Prosobranchia) and *Siphonaria capensis* Q. et G.(Pulmonata). *Journal of Experimental Marine Biology and Ecology*, **128** (3), 191-201.

Meadows, P.S., 1969. Sublittoral fouling communities on northern coasts of Britain. *Hydrobiologia*, **34** (3-4), pp.273-294.

Mieszkowska, N., Burrows, M.T., Pannacciulli, F.G. & Hawkins, S.J., 2014. Multidecadal signals within co-occurring intertidal barnacles *Semibalanus balanoides* and *Chthamalus* spp. linked to the Atlantic Multidecadal Oscillation. *Journal of Marine Systems*, **133**, 70-76.

Miron, G., Bourget, E. & Archambault, P., 1996. Scale of observation and distribution of adult conspecifics: their influence in assessing passive and active settlement mechanisms in the barnacle *Balanus crenatus* (Brugière). *Journal of Experimental Marine Biology and Ecology*, **201** (1), 137-158.

Naylor, E., 1965. Effects of heated effluents upon marine and estuarine organisms. *Advances in Marine Biology*, **3**, 63-103.

Newell, R.C., 1979. *Biology of intertidal animals*. Faversham: Marine Ecological Surveys Ltd.

Newman, W. A. & Ross, A., 1976. Revision of the Balanomorph barnacles including a catalogue of the species. *San Diego Society of Natural History Memoirs*, **9**, 1–108.

OECD (ed.), 1967. *Catalogue of main marine fouling organisms*. Vol. 3: *Serpulids*. Paris: Organisation for Economic Co-operation and Development.

Petraitis, P.S. & Dudgeon, S.R., 2005. Divergent succession and implications for alternative states on rocky intertidal shores. *Journal of Experimental Marine Biology and Ecology*, **326** (1), 14-26.

Petraitis, P.S., Rhile, E.C. & Dudgeon, S., 2003. Survivorship of juvenile barnacles and mussels: spatial dependence and the origin of alternative communities. *Journal of Experimental Marine Biology and Ecology*, **293** (2), 217-236.

Povey, A. & Keough, M.J., 1991. Effects of trampling on plant and animal populations on rocky shores. *Oikos*, **61**: 355-368.

Price, J.H., Irvine, D.E. & Farnham, W.F., 1980. *The shore environment. Volume 2: Ecosystems*. London Academic Press.

Pyefinch, K.A. & Mott, J.C., 1948. The sensitivity of barnacles and their larvae to copper and mercury. *Journal of Experimental Biology*, **25**, 276-298.

Raffaelli, D. & Hawkins, S., 1999. *Intertidal Ecology* 2nd edn.. London: Kluwer Academic Publishers.

Rainbow, P.S., 1987. Heavy metals in barnacles. In *Barnacle biology*. *Crustacean issues 5* (ed. A.J. Southward), 405-417. Rotterdam: A.A. Balkema.

Ribeiro, P.A., Xavier, R., Santos, A.M. & Hawkins, S.J., 2009. Reproductive cycles of four species of *Patella* (Mollusca: Gastropoda) on the northern and central Portuguese coast. *Journal of the Marine Biological Association of the United Kingdom*, **89** (06), 1215-1221.

Riley, K. & Ballerstedt, S., 2005. *Spirobranchus triqueter*. *Marine Life Information Network: Biology and Sensitivity Key Information Sub-programme* [on-line]. Plymouth: Marine Biological Association of the United Kingdom:. [cited 08/01/2016]. Available from: https://www.marlin.ac.uk/species/detail/1794

Rognstad, R.L., Wethey, D.S. & Hilbish, T.J., 2014. Connectivity and population repatriation: limitations of climate and input into the larval pool. *Marine Ecology Progress Series*, **495**, 175-183.

Sanford, E., Bermudez, D., Bertness, M.D. & Gaines, S.D., 1994. Flow, food supply and acorn barnacle population dynamics. *Marine Ecology Progress Series*, **104**, 49-49.

Saunders, R. & Connell, S., 2001. Interactive effects of shade and surface orientation on the recruitment of spirorbid polychaetes. *Austral Ecology*, **26** (1), 109-115.

Sebens, K.P., 1985. Community ecology of vertical rock walls in the Gulf of Maine: small-scale processes and alternative community states. In *The Ecology of Rocky Coasts: essays presented to J.R. Lewis, D.Sc.* (ed. P.G. Moore & R. Seed), pp. 346-371. London: Hodder & Stoughton Ltd.

Sebens, K.P., 1986. Spatial relationships among encrusting marine organisms in the New England subtidal zone. *Ecological Monographs*, **56**, 73-96.

Shanks, A.L. & Wright, W.G., 1986. Adding teeth to wave action- the destructive effects of wave-bourne rocks on intertidal organisms. *Oecologia*, **69** (3), 420-428.

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

Southward, A.J. & Southward, E.C., 1978. Recolonisation of rocky shores in Cornwall after use of toxic dispersants to clean up the *Torrey Canyon* spill. *Journal of the Fisheries Research Board of Canada*, **35**, 682-706.

Southward, A.J., 1955. On the behaviour of barnacles. I. The relation of cirral and other activities to temperature. *Journal of the Marine Biological Association of the United Kingdom*, **34**, 403-432.

Southward, A.J., Hawkins, S.J. & Burrows, M.T., 1995. Seventy years observations of changes in distribution and abundance of zooplankton and intertidal organisms in the western English Channel in relation to rising sea temperature. *Journal of Thermal Biology*, **20**, 127-155.

Stubbings, H.G. & Houghton, D.R., 1964. The ecology of Chichester Harbour, south England, with special reference to some fouling species. *Internationale Revue der Gesamten Hydrobiologie*, **49**, 233-279.

Terry, L. & Sell, D., 1986. Rocky shores in the Moray Firth. *Proceedings of the Royal Society of Edinburgh. Section B. Biological Sciences*, **91**, 169-191.

Thomas, J.G., 1940. *Pomatoceros*, *Sabella* and *Amphitrite*. LMBC Memoirs on typical British marine plants and animals no.33. University Press of Liverpool. Liverpool

Thompson, R.C., Olsen, Y., Mitchell, R.P., Davis, A., Rowland, S.J., John, A.W., McGonigle, D. & Russell, A.E., 2004. Lost at sea: where is all the plastic? *Science*, **304** (5672), 838-838.

Thorp, C., Jones-Knight, P. & Jones-Knight, E., 1986. New records of tubeworms established in British harbours. *Journal of the Marine Biological Association of the United Kingdom*, **66** (04), 881-888.

Tighe-Ford, D., 1967. Possible mechanism for the endocrine control of breeding in a cirripede. *Nature*, **216**, 920-921.

Tillin, H. & Tyler-Walters, H., 2014. Assessing the sensitivity of subtidal sedimentary habitats to pressures associated with marine activities. Phase 2 Report – Literature review and sensitivity assessments for ecological groups for circalittoral and offshore Level 5 biotopes. *JNCC Report* No. 512B, 260 pp. Available from: www.marlin.ac.uk/publications

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

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

Warner, G.F., 1985. Dynamic stability in two contrasting epibenthic communities. In *Proceedings of the 19th European Marine Biology Symposium, Plymouth, Devon, UK, 16-21 September, 1984* (ed. P.E. Gibbs), pp. 401-410.

Watson, D.I., O'Riordan, R.M., Barnes, D.K. & Cross, T., 2005. Temporal and spatial variability in the recruitment of barnacles and the local dominance of *Elminius modestus* Darwin in SW Ireland. *Estuarine, Coastal and Shelf Science*, **63** (1), pp.119-131.

Wethey, D.S., 1985. Catastrophe, Extinction, and Species Diversity: A Rocky Intertidal Example. *Ecology*, **66** (2), 445-456.

Wethey, D.S., 1984. Sun and shade mediate competition in the barnacles *Chthamalus* and *Semibalanus*: a field experiment. *The Biological Bulletin*, **167** (1), 176-185.

Witt, J., Schroeder, A., Knust, R. & Arntz, W.E., 2004. The impact of harbour sludge disposal on benthic macrofauna communities in the Weser estuary. *Helgoland Marine Research*, **58** (2), 117-128.

Witte, S., Buschbaum, C., van Beusekom, J.E. & Reise, K., 2010. Does climatic warming explain why an introduced barnacle finally takes over after a lag of more than 50 years? *Biological Invasions*, **12** (10), 3579-3589.

Wood, C., 2015. The red ripple bryozoan *Watersipora subatra*. *Great Britain Non-native Species Secretariat*. [On-line][cited 16/06/2015]. Available from: http://www.nonnativespecies.org/factsheet/factsheet.cfm?speciesId=3748