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Marine Information Network

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

Sponges and shade-tolerant red seaweeds on overhanging lower eulittoral bedrock and in cave entrances

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

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2016-07-03

A report from:

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

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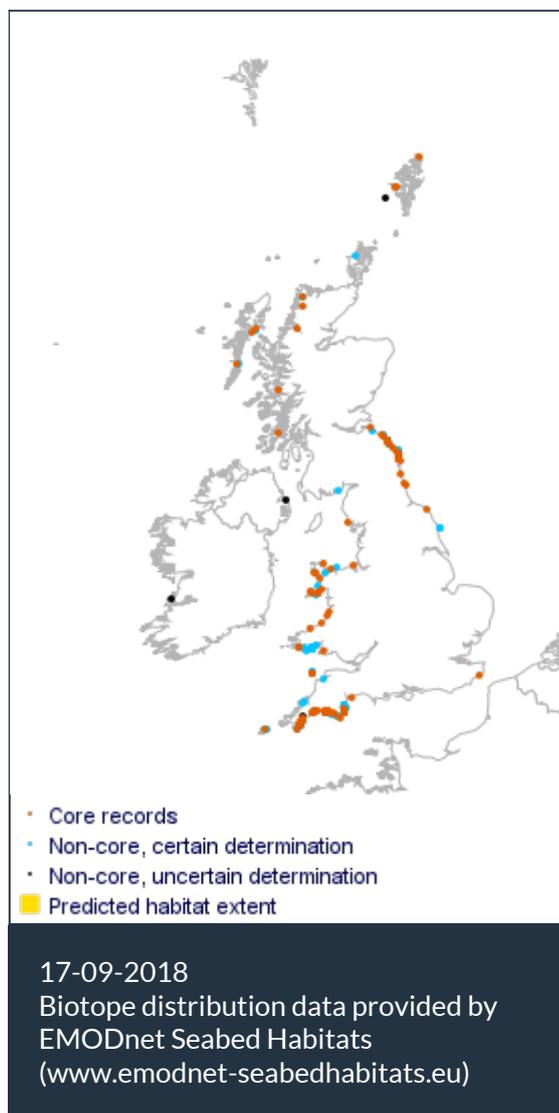
This review can be cited as:

Readman, J.A.J., 2016. Sponges and shade-tolerant red seaweeds on overhanging lower eulittoral bedrock and in cave entrances. In Tyler-Walters H. and Hiscock K. (eds) *Marine Life Information Network: Biology and Sensitivity Key Information Reviews*, [on-line]. Plymouth: Marine Biological Association of the United Kingdom. DOI <https://dx.doi.org/10.17031/marlinhab.61.1>



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Researched by John Readman Refereed by This information is not refereed.

Summary

☰ UK and Ireland classification

EUNIS 2008	A1.446	Sponges and shade-tolerant red seaweeds on overhanging lower eu littoral bedrock and in cave entrances
JNCC 2015	LR.FLR.CvOv.SpR	Sponges and shade-tolerant red seaweeds on overhanging lower eu littoral bedrock and in cave entrances
JNCC 2004	LR.FLR.CvOv.SpR	Sponges and shade-tolerant red seaweeds on overhanging lower eu littoral bedrock and in cave entrances
1997 Biotope	LR._Ov.SR	Sponges and shade-tolerant red seaweeds on overhanging lower eu littoral bedrock

🔍 Description

Overhanging shaded bedrock on the open lower shore and at the entrance to inner reaches of caves (where light availability permits), which is not subject to appreciable wave-surge,

characterized by a shade-tolerant red seaweed community. It includes foliose species such as *Plumaria plumosa*, *Palmaria palmata*, *Mastocarpus stellatus*, *Membranoptera alata* and *Osmundea pinnatifida*, but *Lomentaria articulata* and coralline crusts are usually present as well. The foliose green seaweed *Ulva lactuca* can be present. The rock surface often supports dense populations of calcareous tube-forming polychaetes *Spirorbis* spp. and *Spirobranchus* spp., while sponges such as *Grantia compressa*, *Halichondria panicea* and *Hymeniacion perleve* can be common. The hydroid *Dynamena pumila* (normally found on fucoids) hangs in distinct form from overhanging rock. Colonies of the ascidian *Botryllus schlosseri* can be found on the rock, along with the mussel *Mytilus edulis* and the barnacles *Semibalanus balanoides* and *Perforatus perforatus* (the latter may occur at high densities in the south and west), while the anemone *Actinia equina* thrives in the permanently damp pits and crevices. The whelk *Nucella lapillus* can be found among the barnacles and mussels, preying on them. The long list of characterizing species is partly due to the difference in the species composition and does not solely reflect a high species richness.

↓ Depth range

Lower shore

Additional information

-

✓ Listed By

- none -

Further information sources

Search on:



Sensitivity review

Sensitivity characteristics of the habitat and relevant characteristic species

This group of biotopes occur around cave entrances and overhangs on rock in the littoral zone. These are typically shaded, resulting in limited (or no) algal presence. The group is characterized by the presence of the sponges *Hymeniacidon perleve* and *Halichondria panicea*.

LR.FLR.CvOv.SpR occurs on overhanging shaded bedrock in the lower shore. In addition to the sponges, it is characterized by the presence of shade tolerant red seaweeds including *Plumaria plumosa*, *Palmaria palmata*, *Mastocarpus stellatus*, *Membranoptera alata* and *Osmundea pinnatifida* and coralline crusts.

LR.FLR.CvOv.SpR.Den is a sub-biotope that is found further into caves than LR.FLR.CvOv.SpR (Connor *et al.*, 2004). It is characterized by dense colonies of the ascidian *Dendrodoa grossularia*.

LR.FLR.CvOv.SpByAs is very similar to LR.FLR.CvOv.SpR, but occurs in more shaded overhangs and caves and lacks the shade tolerant red seaweed component.

In conclusion, these biotopes are characterized by the presence of the sponges *Halichondria panicea* and *Hymeniacidon perleve*. Other key species important in defining the biotopes include shade tolerant red seaweeds and the ascidian *Dendrodoa grossularia*. Other characterizing species among the biotopes include barnacles (such as *Balanus crenatus*, *Perforatus perforatus* and *Semibalanus balanoides*, hydroids including *Dynamena pumila* and spirorbids. These species are likely to be ubiquitous and the focus of the assessments are the important characterizing species. However, they are considered where appropriate.

Resilience and recovery rates of habitat

Little information on sponge longevity and resilience exists. Reproduction can be asexual (e.g. budding) or sexual (Naylor, 2011) and individual sponges are usually hermaphrodites (Hayward & Ryland, 1994). Short-lived ciliated larvae are released via the aquiferous system of the sponges and metamorphosis follows settlement. Growth and reproduction are generally seasonal (Hayward & Ryland, 1994). Rejuvenation from fragments is also considered an important form of reproduction in sponges (Fish & Fish, 1996). Some sponges are known to be highly resilient to physical damage with an ability to survive severe damage, regenerate and reorganize to function fully again. However, this recoverability varies between species (Wulff, 2006).

Many sponges recruit annually and growth can be rapid, with a lifespan of one to several years (Ackers, 1983). However, sponge longevity and growth has been described as highly variable depending on the species and environmental conditions (Lancaster *et al.*, 2014). It is likely that erect sponges are generally longer lived and slower growing, given their more complex nature than smaller encrusting or cushion sponges. Marine sponges often harbour dense and diverse microbial communities, which can include bacteria, archaea and single-celled eukaryotes (fungi and microalgae). The microbial community can comprise up to 40% of sponge volume, and may have a profound impact on host biology (Webster & Taylor, 2012).

Fowler & Laffoley (1993) monitored the marine nature reserves in Lundy and the Isles Scilly and found that a number of common sponges showed great variation in size and cover during the study period. Large colonies appeared and vanished at some locations. Some large encrusting sponges

went through periods of both growth and shrinkage, with considerable changes taking place from year to year. For example, *Cliona celata* colonies generally grew extremely rapidly, doubling their size or more each year but, in some years an apparent shrinkage in size took place. In contrast, there were no obvious changes in the cover of certain unidentified thin encrusting sponges.

Hymeniacidon perleve is found in thin sheets, cushions and rarely as erect and branching. It is found from the Arctic to the Mediterranean from the littoral to the circalittoral (Ackers *et al.*, 1992). *Halichondria panicea* is very polymorphic, varying from thin sheets, massive forms and cushions to branching. It crumbles readily and branches are brittle (breaking if bent through 20°). As an opportunistic species, it is found in wide range of niches on rock or any other hard substratum (Ackers *et al.*, 1992). Barthel (1986) reported that *Halichondria panicea* in the Kiel Bight went through annual cycles, with growth occurring between March and July. After July, a strong decline in mean individual weight occurred until the end of September. No change in individual weight was observed over winter, although a change in biochemical composition (condition index and protein, lipid and glycogen content) was noted. Reproductive activity occurred in August and September with young colonies appearing in early autumn. Adult *Halichondria panicea* degenerated and disintegrated after reproduction. Fish & Fish (1996), however, suggested a lifespan of about three years and Vethaak *et al.*, (1982) reported that *Halichondria panicea* survived the winter in a normal, active state in the Oosterschelde. Fell & Lewandrowski (1981) observed the population dynamics of *Halichondria* spp. within an eelgrass bed in a lower estuary in Connecticut over a two year period. Large numbers of larval derived specimens developed on the eelgrass during the summer, and many of these sponges became sexually reproductive, further increasing the size of the population. However, mortality was high, and at the end of the summer, only a relatively small sponge population remained. Sexual reproduction by larva-derived specimens of *Halichondria* spp. occurred primarily after breeding by the parental generation had declined. The larva-derived sponges grew rapidly, and the percentage of specimens containing large, female reproductive elements increased with specimen size. *Halichondria* spp. exhibited an opportunistic life strategy with a 'high rate of turnover'. Gaino *et al.* (2010) observed reproduction within two communities of *Hymeniacidon perlevis* (syn. *Hymeniacidon perleve*). The onset of gametogenesis seemed to be triggered by environmental parameters, amongst which water temperature constituted the most relevant factor statistically. It was reported that differentiation and growth of the sexual elements were asynchronous, with reproduction lasting five months for the females and three months for the males in the Mar Piccolo di Tarant, Italy, from the end of spring to the late summer. Afterwards, the sponges disappeared with no recovery evident up to the end of monitoring (up to late winter 2007).

Thomassen & Riisgard (1995) described a number of studies looking at the growth rates of *Halichondria* spp. with rates of between 1% and 3.3% of total volume per day. Sebens (1985, 1986) monitored recolonization of epifauna on cleared vertical rock walls and described the sponge *Halichondria panicea* as reaching pre-clearance levels of cover after 2 years.

The red algae have complex life histories and exhibit distinct morphological stages over the reproductive life history. Alternation occurs between asexual spore producing stages (tetrasporophytes) and male and female plants producing sexually. Life history stages can be morphologically different or very similar. The tetrasporophyte phase of *Mastocarpus stellatus* is known as the petrocelis and is a flat crust, capable of growing laterally and covering extensive areas. The holdfasts of individual *Chondrus crispus* (a species very similar *Mastocarpus stellatus*) can coalesce over time and can form an extensive crust on rock (Taylor *et al.*, 1981). Other red algae found within the biotope also have life stages that include prostrate creeping bases e.g. the encrusting corallines and *Osmundea pinnatifida*, whereas, in other species such as *Palmaria palmata*,

the thallus or fronds arise from a small discoid holdfast. The basal crusts and crustose tetrasporophytes are perennial, tough, resistant stages that may prevent other species from occupying the rock surface and allow rapid regeneration. They may, therefore, provide a significant recovery mechanism. Where holdfasts and basal crusts are removed, recovery will depend on recolonization via spores. Norton (1992) reviewed dispersal by macroalgae and concluded that dispersal potential is highly variable, recruitment usually occurs on a much more local scale, typically within 10 m of the parent plant. Hence, it is expected that the algal turf would normally rely on recruitment from local individuals and that recovery of populations via spore settlement, where adults are removed, could be protracted. Minchinton *et al.* (1997) documented the recovery of *Chondrus crispus* after a rocky shore in Nova Scotia, Canada, was totally denuded by an ice scouring event. Initial recolonization was dominated by diatoms and ephemeral macroalgae, followed by fucoids and then perennial red seaweeds. After two years, *Chondrus crispus* had re-established approximately 50% cover on the lower shore and after five years it was the dominant macroalga at this height, with approximately 100% cover. In kelp, canopy removal experiments in the Isle of Man, Hawkins & Harkin (1985) observed a rapid increase in the number of *Palmaria palmata* sporelings and the species came to dominate cleared plots within five months. Rhodophyceae have non-flagellate, and non-motile spores that stick on contact with the substratum. Norton (1992) noted that algal spore dispersal is probably determined by currents and turbulent deposition. However, red algae produce large numbers of spores that may settle close to the adult especially where currents are reduced by an algal turf or in kelp forests. It is likely that this species could recolonize an area from adjacent populations within a short period of time in ideal conditions. However, since the dispersal range of spores is limited because the female does not release carpospores and needs to be close to the adult male population, recolonization from distant populations would probably take much longer.

Dendrodoa grossularia is a small solitary ascidian (1.5-2 cm diameter (Millar, 1954)). Settlement occurs from April-June, individuals reach their maximum size by the following summer. Life expectancy is expected to be 18-24 months. Sexual maturity is reached within the second year of growth and the release of gametes occurs from spring-autumn, with peaks in early spring and another in late summer. Gamete release is reduced at temperatures above 15°C and totally suppressed above ca. 20°C (Millar, 1954). Kenny & Rees (1994) observed *Dendrodoa grossularia* was able to recolonize rapidly following aggregate dredging. Following experimental dredging of a site off the English coast, which extracted an area of 1-2 m wide and 0.3-0.5 m deep, *Dendrodoa grossularia* was able to recolonize and attained 40% of pre-dredge abundance and 23% of biomass within 8 months. This recovery rate combined with the ability of this species to reach sexual maturity within its first year, suggests that *Dendrodoa grossularia* can recover from disturbance events within 2 years.

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 three months for these species, based on the 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.

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 the 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 *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, the 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 balanoides* dying within ten days (Kendall *et al.*, 1985). Presumably, these factors also influence the transport, supply and settlement of *Chthamalus* spp., *Balanus crenatus* and other species such as spirorbids that produce pelagic larvae.

The recolonization of epifauna on vertical rock walls was investigated by Sebens (1985, 1986). He reported that rapid colonizers such as encrusting corallines, encrusting bryozoans, amphipods and tubeworms recolonized within 1-4 months. Ascidians such as *Dendrodoa carnea*, *Molgula manhattensis* and *Aplidium* spp. achieved significant cover in less than a year, and, together with *Halichondria panicea*, reached pre-clearance levels of cover after two years.

Resilience assessment

It should be noted that the biotope is characterized by the presence of shade resistant red algae and coralline crusts. However these are likely to be present in an impoverished form and recovery to pre-disturbance abundance is, therefore, likely to be more rapid than in biotopes with a richer abundance of red seaweeds. Whilst some species may take more than two years to completely recover following clearance, as the biotope is defined by a range of shade tolerant red seaweeds, more resilient characterizing species would probably achieve pre-clearance abundance rapidly, such as *Palmaria palmata* (Hawkins & Harkin, 1985), and the biotope classification would be preserved. Sebens (1985, 1986) found that ascidians, encrusting bryozoans and tube worms all achieved significant cover in less than a year following complete removal, and, together with *Halichondria panicea*, reached pre-clearance levels of cover after two years. *Dendrodoa grossularia*

can reach sexual maturity within 1 year of growth and can recover rapidly following severe habitat alteration. Overall, resilience is assessed as 'High' (within two 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)	Medium Q: High A: Medium C: Medium	High Q: High A: Medium C: Medium	Low Q: High A: Medium C: Medium

Intertidal species are exposed to extremes of high and low air temperatures during periods of emersion. They must also be able to cope with sharp temperature fluctuations over a short period 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 this intertidal biotope are therefore generally adapted to tolerate a range of temperatures.

The characterizing red algal species found in this biotope have a range of geographic distributions with some having a 'southern distribution' with their range encompassing warmer waters and others having a 'northern' distribution. Temperature tolerances are therefore likely to vary between species so that long-term changes in temperature have the potential to shift the species composition of this biotope to one more suited to the prevailing thermal regime. Populations can acclimate to prevailing conditions (Davison, 1991; Lohrmann *et al.*, 2004) that can alter tolerance thresholds and care should, therefore, be used when interpreting reported tolerances.

Palmaria palmata does well in low temperatures, with an optimum between 6 and 15°C (Bak, 2014), consistent with a distribution in northern temperate and arctic waters. The species is also found in warmer temperate waters as far south as Portugal in Europe and with localized large populations in northern Spain (Garbary *et al.*, 2012). In the laboratory, plants only became fertile if left at temperatures between 5-7°C with a short light period (Van der Meer & Chen, 1979).

Temperatures at or above 15°C may induce physiological stress (Werner & Dring, 2011; Morgan *et al.*, 1980). In tank cultures of *Palmaria palmata* maintained at 20°C and above, all plants were dead within a week (Morgan *et al.*, 1980). Kain & Norton (1990) suggest, however, that a widely distributed species like *Palmaria palmata* reacts less strongly to temperature differences than some other red algae.

Corallina officinalis also has a cosmopolitan distribution (Guiry & Guiry, 2015) and may tolerate between minus 4 to 28°C (Lüning, 1990). Abrupt temperature changes, e.g. 10°C in California (Seapy & Littler, 1982) and 4.8 to 8.5°C (Hawkins & Hartnoll, 1985) resulted in dramatic declines.

However, in both cases recovery was rapid, suggesting that the crustose bases survived. Latham (2008) investigated the effects of temperature stress on *Corallina officinalis* through laboratory tests on samples collected in the autumn in Devon, England from rockpools. Samples were kept at 15°C for three days and then exposed to temperatures of 5°C, 15°C, 20°C, 25°C and 30°C. The normal range of temperature experienced was suggested to be between 5 and 15°C. At 35°C the *Corallina* was completely bleached after 3 days with the sample kept at 30°C beginning to bleach. After 7 days (the end of the experiment) the sample kept at 30°C was partially bleached while samples kept at lower temperatures showed little change in chemicals produced in reaction to thermal stress and no bleaching, suggesting the temperatures in that range had not induced stress reactions.

The characterizing sponges *Halichondria panacea* and *Hymeniacion perleve* are widely distributed across the coasts of the British Isles and are all found from the Channel Isles to northern Scotland (NBN, 2015). Berman *et al.* (2013) monitored sponge communities off Skomer Island, UK, over three years with all characterizing sponges for this biotope assessed. Seawater temperature, turbidity, photosynthetically active radiation and wind speed were all recorded during the study. It was concluded that, despite changes in species composition, primarily driven by the non-characterizing *Hymeraphia*, *Stellifera* and *Halicnemis patera*, no significant difference in sponge density was recorded in all sites studied. Morphological changes most strongly correlated with a mixture of water visibility and temperature. Lemoine *et al.* (2007) studied the effects of thermal stress on the holobiont of the sponge *Halichondria bowerbanki* collected from Virginia, USA. Whilst no apparent change in density or diversity of symbionts was detected over the range of temperatures (29°C, 30°C and 31°C), the presence of particular symbionts was temperature dependent. Barthel (1986) reported that reproduction and growth in *Halichondria panacea* in the Kiel Bight were primarily driven by temperature, with higher temperatures corresponding with the highest growth.

For the ascidian *Dendrodoa grossularia*, gamete release occurs from spring-autumn, with peaks in early spring and another in late summer. Gamete release is reduced at temperatures above 15°C and totally suppressed above 20°C (Millar, 1954). No information was found on the upper temperature threshold of mature *Dendrodoa grossularia*. Whilst widespread throughout the British Isles (NBN, 2015), *Dendrodoa grossularia* is close to its southern limit and a dramatic increase in temperature outside the normal range for the UK may cause mortality.

The distribution of the key characterizing species, *Semibalanus balanoides* is 'northern' with their range extending to the Arctic circle. Populations in the southern part of England are relatively close to the southern edge of their geographic range. 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). Increased temperatures are likely to favour chthamalid barnacles or *Austrominius modestus* in the sheltered variable salinity biotopes rather than *Semibalanus balanoides* (Southward *et al.*, 1995).

Sensitivity assessment

The 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. However, an acute high temperature event in the summer would approach the upper tolerance of some species and may result in some mortality. Overall, resistance is assessed as 'Medium', resilience is, therefore, 'High' and sensitivity of the biotope is 'Low'.

Temperature decrease (local)

Medium

Q: Medium A: Medium C: Medium

High

Q: High A: Medium C: Medium

Low

Q: Medium A: Medium C: Medium

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 (Davenport & Davenport, 2005).

Palmaria palmata does well in low temperatures, with an optimum between 6 and 15°C, consistent with a distribution in northern temperate and arctic waters. This species is also found in warmer temperate waters as far south as Portugal in Europe and with localized large populations in northern Spain (Garbary *et al.*, 2012). In the laboratory, plants only became fertile if left at temperatures between 5-7°C with a short light period (Van der Meer & Chen, 1979). Acute or chronic changes in temperature below 5°C may, therefore, reduce reproductive success, although reproduction and vegetative growth in warmer months should compensate for any reduction in output. Temperatures at or above 15°C may induce physiological stress (Werner & Dring, 2011; Morgan *et al.*, 1980). In tank cultures of the species at 20°C and above, all plants were dead within a week (Morgan *et al.*, 1980). Such high temperatures, however, are unlikely in most parts of Britain and Ireland. The growth of *Palmaria palmata* is reduced below 6°C but the plant can survive lower temperatures as long as the tissue does not freeze (Werner & Dring, 2011). Lüning (1990) reported that *Corallina officinalis* from Helgoland survived 0°C when exposed for one week. New Zealand specimens were found to tolerate -4°C (Frazer *et al.*, 1988).

The characterizing sponges *Halichondria panacea* and *Hymeniacidon perleve* are widely distributed across the coasts of the British Isles and are found from the Channel Isles to northern Scotland (NBN, 2015). Berman *et al.* (2013) monitored sponge communities off Skoma Island, UK, over three years with all characterizing sponges for this biotope assessed. Seawater temperature, turbidity, photosynthetically active radiation and wind speed were all recorded during the study. It was concluded that, despite changes in species composition, primarily driven by the non-characterizing *Hymeraphia Stellifera* and *Halicnemia patera*, no significant difference in sponge density was recorded in all sites studied. Morphological changes most strongly correlated with a mixture of visibility and temperature.

Crisp (1964) studied the effects of an unusually cold winter (1962-63) on the marine life in Britain, including porifera in north Wales. Whilst difficulty in distinguishing between mortality and delayed development was noted, Crisp found that *Halichondria panicea* was wholly or partly killed by frost and *Hymeniacidon perleve* was unusually rare. Barthel (1986) reported that *Halichondria panicea* in the Kiel Bight degenerated and disintegrated after reproduction before winter,

however, young colonies were observed from September. However, *Dendrodoa grossularia* has been recorded as an abundant component of benthic fauna in Nottinghambukta, Svalbard (Różycki & Gruszczyński, 1991).

Sensitivity assessment

There is evidence of sponge mortality at extreme low temperatures in the British Isles. Given this evidence, it is likely that a cooling of 5°C for a month could affect the characterizing sponges, and resistance has been assessed as '**Medium**' with a resilience of '**High**'. Sensitivity has, therefore, been assessed as '**Low**' at the benchmark level.

Salinity increase (local)

Low

Q: Low A: NR C: NR

High

Q: High A: Medium C: Medium

Low

Q: Low A: Low C: Low

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.

Laboratory experiments have defined the upper and lethal lower limits for *Palmaria palmata* as 15 psu and 50 psu (Karsten *et al.*, 2003), with optimal salinity defined as 23-34 psu (Robbins, 1978). It is not clear whether a sustained increase in salinity would be tolerated, or whether different life stages have different tolerances as this species is under studied in comparison to some other species of commercial importance (Bak, 2014). The characterizing crustose corallines and *Corallina officinalis* are found in rockpools where salinities may fluctuate markedly during exposure to the air. Kinne (1971b) cited maximal growth rates for *Corallina officinalis* between 33 and 38 psu in Texan lagoons. 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).

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).

No evidence for *Halichondria panicea* in hypersaline conditions was found.

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 increase in salinity to > 40 ppt. Resistance is therefore assessed as '**Low**' and recovery as '**High**'. Sensitivity is therefore assessed as '**Low**'.

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

The characterizing sponge species *Halichondria panicea* and *Hymeniacion perleve* occur in harbours and estuaries (Ackers *et al.*, 1992) and have both been recorded in biotopes of variable salinity (Connor *et al.*, 2004). *Dendrodoa grossularia* has been recorded as an abundant component of benthic fauna in Nottinghambukta, Svalbard where salinity can range between 6 and 20‰ (Różycki & Gruszczyński, 1991).

Laboratory experiments have defined the upper and lethal lower limits for *Palmaria palmata* as 15 psu and 50 psu, (Karsten *et al.*, 2003) with optimal salinity defined as 23-34 psu (Robbins, 1978). High growth rates of *Palmaria palmata* were observed in cultivation experiments at 21 psu in the *in-situ* *Palmaria palmata* from the Arctic Kongsfjord (Spitsbergen) exposed for four days to salinities of 15 psu, following freshwater run-off suffered high levels of mortality (Karsten *et al.*, 2003). In Danish waters where salinities are lowered by freshwater inputs, cultivation experiments found that *Palmaria palmata* growth over a seven month period was greatest at a site where mean salinity was 21±3 psu, while no growth occurred at a site where mean salinity was 17±5 psu (Bak, 2014).

Barnacles 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". In this state, motor activity ceases and respiration falls, enabling animals to survive in freshwater for three weeks (Barnes, 1953). *Semibalanus balanoides* can tolerate salinities down to 12 psu, below which cirral activity ceases (Foster, 1970).

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

The majority of the species in the biotope have been recorded in areas of low salinity (such as estuaries). Although the overall nature of the biotope would probably be unaffected, some species, such as *Palmaria palmata*, are likely to experience some mortality. Resistance is, therefore, '**Medium**', resilience is '**High**' and sensitivity is '**Low**'.

Water flow (tidal current) changes (local) **High** **High** **Not sensitive**
 Q: Low A: NR C: NR Q: High A: High C: High Q: Low A: Low C: Low

Riisgard *et al.* (1993) discussed the low energy cost of filtration for sponges and concluded that passive current-induced filtration may be insignificant for sponges. Pumping and filtering occurs in choanocyte cells that generate water currents in sponges using flagella (De Vos *et al.*, 1991). The biotopes assessed occur in moderate to negligible tidal flows (0-1.5 m/sec). These biotopes occur on overhangs and caves that are partially sheltered from direct wave action, in moderately wave exposed to wave sheltered shores (Connor *et al.*, 2004). Changes in wave action will probably affect the degree of splash and spray, and, hence, moisture, desiccation and salinity experienced by the biotopes.

Sensitivity assessment

All the biotopes within this complex are characterized by their moderate energy and a significant increase or decrease would probably result in a fundamental change to the nature of the biotope and, hence, reclassification would be required. However, at the benchmark level, changes are unlikely to be significant enough to alter the biotope and resistance is, therefore, assessed as 'High', resilience is 'High' and the biotope is 'Not sensitive' at the benchmark level.

Emergence regime changes

Low

Q: Medium A: Medium C: Medium

High

Q: High A: Medium C: Medium

Low

Q: Medium A: Medium 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 desiccation and reduced feeding opportunities for filter feeders including spirobids, barnacles, sponges and anemones which feed when immersed.

Semibalanus balanoides is less tolerant of desiccation stress than *Chthamalus* barnacle 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. It should be noted that moisture from wave surge is considered important in maintaining faunal abundance (Connor *et al.*, 2004). Changes in emergence may therefore eventually lead to replacement of this biotope to one more tolerant of desiccation.

Decreased emergence would reduce desiccation 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). Mobile species present within the biotope 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.ScrFa.

Corallina officinalis may occur at a range of shore heights depending on local conditions such as the degree of wave action (Dommasnes, 1969), shore topography, run-off and degree of shelter from canopy forming macroalgae. At Hinkley Point (Somerset, England), for example, seawater run-off from deep pools high in the intertidal supports dense turfs of *Corallina* spp. lower on the shore (Bamber & Irving, 1993). Fronds are highly intolerant of desiccation and do not recover from a 15% water loss, which might occur within 40-45 minutes during a spring tide in summer (Wiedemann, 1994). Bleached corallines were observed 15 months after the 1964 Alaska earthquake which elevated areas in Prince William Sound by 10 m. Similarly, increased exposure to air caused by upward movement of 15 cm due to nuclear tests at Armchitka Island,

Alaska, adversely affected *Corallina pilulifera* (Johansen, 1974). During an unusually hot summer, Hawkins & Hartnoll (1985) observed damaged *Corallina officinalis* and other red algae. Littler & Littler (1984) suggested that the basal crustose stage is adaptive, allowing individuals to survive periods of physical stress as well as physiological stress such as desiccation and heating. The basal crust stage may persist for extended periods with frond regrowth occurring when conditions are favourable.

Sensitivity assessment

This biotope occurs from the mid-littoral to the sublittoral fringe. Increased emergence may reduce habitat suitability for characterizing species through greater exposure to desiccation and reduced feeding opportunities for the faunal community while decreased emergence would reduce desiccation stress and allow the attached epifauna more feeding time, although may result in succession. As emergence is a key factor structuring the distribution of species 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: Low A: NR C: NR

High

Q: High A: High C: High

Not sensitive

Q: Low A: Low C: Low

These biotopes occur on overhangs and caves that are partially sheltered from direct wave action, in moderately wave exposed to wave sheltered shores (Connor *et al.*, 2004). Changes in wave action will probably affect the degree of splash and spray, and hence moisture, desiccation and salinity experienced by the biotopes. Therefore, a change in wave exposure will probably affect the upper and lower extent of the biotopes, and result in loss of extent. The effects of wave surge considered to be one aspect in differentiating between SpR.Den and SpByAs biotopes. Increased wave height could result in increased scour which may result in a decline in abundance of the characterizing species.

Sensitivity assessment

Whilst this biotope is recorded as occurring from exposed to sheltered conditions, the local conditions are likely to be an important factor in assessing effects. These biotopes are found under overhangs and on the walls of caves which may afford some protection to the characterizing species. A significant change in wave exposure would probably result in a fundamental change to the nature of the biotope. However, at the benchmark level, changes are unlikely to be significant and resistance is therefore assessed as '**High**', resilience is '**High**' and the biotope is '**Not sensitive**' at the benchmark level.

Chemical Pressures

Transition elements & organo-metal contamination

Resistance

Not Assessed (NA)

Q: NR A: NR C: NR

Resilience

Not assessed (NA)

Q: NR A: NR C: NR

Sensitivity

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). Total growth inhibition was caused by 1 ppm Hg.

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).

While some sponges, such as *Cliona* spp. have been used to monitor heavy metals by looking at the associated bacterial community (Marques *et al.*, 2007; Bauvais *et al.*, 2015), no literature on the effects of transition element or organo-metal pollutants on the characterizing sponges could be found.

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.

Tethya lyncurium concentrated BaP (benzo[a]pyrene) to 40 times the external concentration and no significant repair of DNA was observed in the sponges, which, in higher animals, would likely lead to cancers. As sponge cells are not organized into organs the long-term effects are uncertain (Zahn *et al.*, 1981). No information was found on the intolerance of the characterizing sponges or barnacles 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).

Ignatiades & Becacos-Kontos (1970) found that *Ciona intestinalis* can resist the toxicity of oil polluted water and ascidians are frequently found in polluted habitats such as marinas and harbours, etc. (Carver *et al.*, 2006) as well as *Ascidia mentula* (Aneiros *et al.*, 2015).

Ryland & De Putron (1998) found no detectable damage to under-boulder communities, which are similar to some overhang communities, in Watwick Bay, Pembrokeshire following the *Sea Empress* oil spill. Part of the resistance to effects might be because oil does not settle onto overhanging surfaces. However, some species, especially gastropods are likely to be narcotised and killed and some damage is likely. Return to a previous species composition would occur from new settlement from larval sources although some gastropods have no or only a short dispersal phase so that recovery will be slow.

O'Brien & Dixon (1976) suggested that red algae were the most sensitive group of algae to oil or dispersant contamination, possibly due to the susceptibility of phycoerythrins to destruction, but that the filamentous forms were the most sensitive. Laboratory studies of the effects of oil and dispersants on several red algae species, including *Palmaria palmata* (Grandy, 1984 cited in Holt *et al.*, 1995) concluded that they were all sensitive to oil/ dispersant mixtures, with little differences between adults, sporelings, diploid or haploid life stages.

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, dependent on the concentration and type of chemical involved (Holt *et al.*, 1995). *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.

Cole *et al.* (1999) suggested that herbicides, such as simazina and atrazine were very toxic to macrophytes. Hoare & Hiscock (1974) noted that all red algae were excluded from Amlwch Bay, Anglesey, by acidified halogenated effluent discharge. Such evidence suggests *Palmaria palmata* has a high intolerance to synthetic chemicals at levels greater than the benchmark.

Radionuclide contamination

No evidence (NEv)

Q: NR A: NR C: NR

Not relevant (NR)

Q: NR A: NR C: NR

No evidence (NEv)

Q: NR A: NR C: NR

'No evidence' was found.

Introduction of other substances

Not Assessed (NA)

Q: NR A: NR C: NR

Not assessed (NA)

Q: NR A: NR C: NR

Not assessed (NA)

Q: NR A: NR C: NR

This pressure is **Not assessed**.

De-oxygenation

High

Q: Medium A: Low C: Medium

High

Q: High A: High C: High

Not sensitive

Q: Medium A: Low C: Medium

It is likely that as this biotope occurs in areas that are shallow and tidally flushed where 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, respire 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).

Halichondria panicea has been reported to survive under oxygen levels as low as 0.5-4% saturation (ca 0.05-0.4 mg/l) for up to 10 days (Mills *et al.*, 2014).

Little information on the effects of oxygen depletion on macroalgae was found, although Kinne (1972) reports that reduced oxygen concentrations inhibit both photosynthesis and respiration which may affect growth and reproduction. The effects of decreased oxygen concentration

equivalent of the benchmark would be greatest during the dark when the macroalgae are dependent on respiration. However, this biotope occurs in the intertidal and *Palmaria palmata* will be able to respire during periods of emersion.

Sensitivity assessment

Based on evidence for the characterizing *Semibalanus balanoides*, *Halichondria panicea* and considering 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' at the benchmark level.

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

The benchmark is set at compliance with WFD criteria for good status, based on nitrogen concentration (UKTAG, 2014). 'Not sensitive' at the pressure benchmark that assumes compliance with good status as defined by the WFD.

This pressure relates to increased levels of nitrogen, phosphorus and silicon in the marine environment compared to background concentrations. The nutrient enrichment of a marine environment leads to organisms no longer being limited by the availability of certain nutrients. The consequent changes in ecosystem functions can lead to the progression of eutrophic symptoms (Bricker *et al.*, 2008), changes in species diversity and evenness (Johnston & Roberts, 2009) decreases in dissolved oxygen and uncharacteristic microalgae blooms (Bricker *et al.*, 1999, 2008).

Johnston & Roberts (2009) undertook a review and meta-analysis of the effect of contaminants on species richness and evenness in the marine environment. Of the 47 papers reviewed relating to nutrients as a contaminant, over 75% found that it had a negative impact on species diversity, <5% found increased diversity, and the remaining papers finding no detectable effect. Due to the 'remarkably consistent' effect of marine pollutants on species diversity this finding is relevant to this biotope. It was found that any single pollutant reduced species richness by 30-50% within any of the marine habitats considered. These examples were almost entirely from the introduction of nutrients, either from aquaculture or sewage outfalls (Johnston & Roberts, 2009).

Moderate nutrient enrichment, especially in the form of organic particulates and dissolved organic material, is likely to increase food availability for all the suspension feeders within the biotope. However, long-term or high levels of organic enrichment may result in eutrophication and have indirect adverse effects, such as increased turbidity, increased suspended sediment, increased risk of deoxygenation and the risk of algal blooms.

Organic enrichment	High	High	Not sensitive
	Q: Low A: NR C: NR	Q: High A: High C: High	Q: Low A: Low C: Low

Organic enrichment leads to organisms no longer being limited by the availability of organic carbon. The consequent changes in ecosystem function can lead to the progression of eutrophic symptoms (Bricker *et al.*, 2008), changes in species diversity and evenness (Johnston & Roberts, 2009) and decreases in dissolved oxygen and uncharacteristic microalgae blooms (Bricker *et al.*, 1999, 2008). Indirect adverse effects associated with organic enrichment include increased

turbidity, increased suspended sediment and the increased risk of deoxygenation.

Johnston & Roberts (2009) undertook a review and meta-analysis of the effect of contaminants on species richness and evenness in the marine environment. Of the 49 papers reviewed relating to sewage as a contaminant, over 70% found that it had a negative impact on species diversity, <5% found increased diversity, and the remaining papers finding no detectable effect. It was found that any single pollutant reduced species richness by 30-50% within any of the marine habitats considered (Johnston & Roberts, 2009). Throughout their investigation there were only a few examples where species richness was increased due to the anthropogenic introduction of a contaminant. These examples were almost entirely from the introduction of nutrients, either from aquaculture or sewage outfalls.

The animals found within the biotope may be able to utilize 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 utilize 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.

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).

Halichondria occur in harbours and estuaries (Ackers *et al.*, 1992) and may, therefore, tolerate high levels of organic carbon, although no specific evidence for this species was found, other sponges have been described in organically enriched environments. Fu *et al.* (2007) described *Hymeniacion perleve* in aquaculture ecosystems in sterilized natural seawater with different concentrations of total organic carbon (TOC). At several concentrations between 52.9 and 335.13 mg/l, *Hymeniacion perleve* removed 44–61% TOC during 24 h, with retention rates of ca 0.19–1.06 mg/hr per g-fresh sponge. *Hymeniacion perleve* removed organic carbon excreted by *Fugu rubripes* with similar retention rates of ca 0.15 mg/h/ per g-fresh sponge, and the sponge biomass increased by 22.8%.

The crusting coralline *Lithophyllum incrustans* was present at sites dominated by *Ulva* spp. in the Mediterranean exposed to high levels of organic pollution from domestic sewage (Arévalo *et al.*, 2007). As turfs of red algae can trap large amounts of sediment the red algae are not considered sensitive to the sedimentation element of this pressure. Within trapped sediments associated species and deposit feeders would be able to consume inputs of organic matter.

Sensitivity assessment. It is not clear whether the pressure benchmark would lead to enrichment effects in this dynamic habitat. Water movement would disperse organic matter particles, mitigating the effect of this pressure. Based on the AMBI categorization (Borja *et al.*, 2000, Gittenberger & Van Loon, 2011). 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, assessed as 'Not sensitive'.

A 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
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If rock were replaced with sediment, this would represent a fundamental change to the physical character of the biotope and the species would be unlikely to recover. The biotope would be lost. Resistance to the pressure is considered '**None**', and resilience '**Very low**'. Sensitivity has been assessed as '**High**'.

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
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'**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
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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: Medium A: Medium C: Medium	High Q: High A: Medium C: Medium	Low Q: Medium A: Medium C: Medium
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The species characterizing this biotope occur on the rock and, therefore, have no protection from surface abrasion. For this assessment, the focus is on the effects of trampling and scour. *Halichondria panicea* is compressible but crumbly in texture and easily broken (Ackers *et al*, 1992) and is typically found in cryptic or semi-cryptic areas (Hayward & Ryland, 1995b). Abrasion events are therefore likely to remove the sponge, however, specific evidence was not found.

Little information is available on the effects of abrasion on intertidal red algae. Brosnan & Crumrine (1994) found that the foliose red algae *Mastocarpus papillatus* was intolerant of moderate levels of trampling, but recovered rapidly. It should be noted that algal turf increased in abundance following trampling, although this may be due to reduced competition with barnacles.

The effects of trampling (a source of abrasion) on barnacles appear 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 & 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 shoreline /h) 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, Victoria, Australia, found that in single step experiments 10 out of 67 barnacles, (*Chthamalus antennatus* about 3 mm 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 *Cellana 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 the 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.

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 & 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 an 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, however, persistent abrasion from scouring could result in a change to the similar biotope LR.FLR.CvOv.ScrFa (Connor *et al.*, 2004). Resistance is assessed as 'Low' and recovery is 'High'. Sensitivity is assessed as 'Low'.

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. This pressure is 'Not relevant' to hard rock biotopes.

Changes in suspended solids (water clarity)

Medium

Q: Medium A: Medium C: Medium

High

Q: High A: Medium C: Medium

Low

Q: Medium A: Medium C: Medium

Intertidal biotopes will only be exposed to this pressure when submerged during the tidal cycle and thus may have limited exposure. Siltation, which may be associated with increased suspended solids and the subsequent deposition of these is assessed separately (see siltation pressures). In general, increased suspended particles reduce light penetration and increase scour and deposition. They may enhance food supply to filter or deposit feeders (where the particles are organic in origin) or decrease feeding efficiency (where the particles are inorganic and require greater filtration efforts).

Increases in the cover of robust, sediment trapping, turf-forming algae at the expense of canopy forming species have been observed worldwide in temperate systems and has been linked to increased suspended solids linked to human activities worldwide (Airoldi, 2003).

Red algae, in general, are shade tolerant and are common components of the understory on seaweed dominated shores. *Palmaria palmata* is often found under partially shaded conditions as an epiphyte on the stems of *Laminaria* spp. (Morgan *et al.*, 1980) in the sublittoral zone (Lüning, 1990). Research on light tolerance of *Palmaria palmata* is limited (Bak, 2014), however, red algae are well adapted to low light conditions, but may be close to their limit resulting in loss of LR.FLR.CvOv.SpR to LR.FLR.CvOv.SpR.Den or LR.FLR.CvOv.SpByAs. In the Bay of Fundy where the tidal flux of nutrients from the marshes includes a high level of suspended sediment, *Palmaria palmata* grows well despite high turbidity. Irvine (1983), observed morphological adaptation of the plant in fairly sheltered, silty conditions; sometimes the blade divisions are wedge-shaped and finely dissected above or the blade has numerous linear divisions throughout. It is likely that this form reduces possible smothering that may result from increased siltation resulting from increased levels of suspended sediments. In the absence of nutrients, short-term increase in turbidity may affect growth and reproduction, however, as a perennial, the adults will probably survive. *Palmaria palmata* has relatively thin fronds and may be damaged through scour where levels of suspended sediments are high. On sites affected by high levels of resuspended colliery waste particles, Hyslop *et al.* (1997) found that *Palmaria palmata* was reduced or absent, whereas the fucoids were less affected. It is not clear how the levels of suspended solids experienced by these sites relate to the pressure benchmark.

Despite increased sediment being considered to have a negative impact on suspension feeders (Gerrodette & Flechsig, 1979), many encrusting sponges appear to be able to survive in highly sediment conditions (Schönberg, 2015; Bell & Barnes, 2000; Bell & Smith, 2004). Some of the characterizing species occur in harbours and estuaries, including *Halichondria* spp. and *Hymeniacion perleve* (Ackers *et al.*, 1992) which are likely to experience high turbidity events. *Halichondria panicea* has a cleaning mechanism sloughing off its complete outer tissue layer together with any debris (Barthel & Wolfrath, 1989). There is, however, an energetic cost in cleaning resulting in reduced growth. For short-lived species, such as the star ascidian *Botryllus schlosseri*, reduced growth could prove fatal.

Available evidence indicates that *Spirobranchus triqueter* is tolerant of a wide range of suspended sediment concentrations (Riley & Ballerstedt, 2005). Stubbings & 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).

Sensitivity assessment. An increase in suspended sediment could result in scour due to wave action and tidal flow. Overall, resistance is assessed as 'Medium', mainly due to the scour effect as the sediment is removed. Resilience is 'High' and sensitivity is 'Low'.

Smothering and siltation rate changes (light)

Medium

Q: Medium A: Medium C: Medium

High

Q: High A: Medium C: Medium

Low

Q: Medium A: Medium C: Medium

This biotope occurs under overhangs and at the entrance to caves and the majority of the examples of this biotope are therefore unlikely to be affected by an increase in sediment deposition. This being said, where the biotope does occur on the shallow or flat lower surfaces at cave entrances or are adjacent on the surface, mortality could occur, depending on the level and rate of removal of the sediment.

Despite sediment being considered to have a negative impact on suspension feeders (Gerrodette & Flechsig, 1979), many encrusting sponges appear to be able to survive in highly sedimented conditions, and in fact, many species prefer such habitats (Schönberg, 2015; Bell & Barnes, 2000; Bell & Smith, 2004). Whilst complete smothering may result in loss of the sponges, the biotope often occurs on cave walls and ceilings and thus burial is unlikely.

In a review of the effects of sedimentation on rocky coast assemblages, Airoidi (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 (Airoidi, 2003). Crustose corallines have been reported to survive under a turf of filamentous algae and sediment for 58 days (the duration of the 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 cm² 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.

No empirical evidence was found to assess the sensitivity of the key characterizing species, *Palmaria palmata*, to this pressure. Irvine (1983), observed morphological adaptation of the plant in fairly sheltered, silty conditions; sometimes the blade divisions are wedge-shaped and finely dissected above, or the blade has numerous linear divisions throughout. It is likely that this form reduces possible smothering that may result from increased siltation. However, this adaptation occurs in areas with chronic, relatively low levels of siltation and the blade form is unlikely to confer different levels of resistance to a single deposition event at the pressure benchmark. The perennial holdfasts may survive short periods of siltation, even though fronds may be damaged or removed aiding recovery. *Dendrodoa grossularia* is a small ascidian, capable of reaching a size of approx. 8.5 mm (Millar, 1954) and is, therefore, likely to be inundated by

deposition of 5 cm of sediment. If inundation is long lasting then the understory community may be adversely affected.

Sensitivity assessment. Overall, the biotope is unlikely to be affected by smothering, unless adjacent to the sediment surface. Increase in sediment could result in scour following suspension due to wave and tidal flow. Removal of the sediment is likely to be rapid. Overall, resistance is assessed as 'Medium', mainly due to the scour effect as the sediment is removed. Resilience is 'High' and sensitivity is 'Low'.

Smothering and siltation rate changes (heavy)

Medium

Q: Medium A: Medium C: Medium

High

Q: High A: Medium C: Medium

Low

Q: Medium A: Medium C: Medium

This biotope occurs under overhangs and at the entrance to caves and the majority of the examples of this biotope are therefore unlikely to be affected by an increase in sediment deposition. This being said, where the biotope does occur on the shallow or flat lower surfaces at cave entrances, mortality could occur, depending on the level and rate of removal of the sediment.

Despite sediment being considered to have a negative impact on suspension feeders (Gerrodette & Flechsig 1979), many encrusting sponges appear to be able to survive in highly sedimented conditions, and in fact, many species prefer such habitats (Schönberg, 2015; Bell & Barnes 2000; Bell & Smith, 2004). Whilst complete smothering may result in loss of the sponges, the biotope often occurs on cave walls and ceilings and thus burial is unlikely.

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). Crustose corallines have been reported to survive under a turf of filamentous algae and sediment for 58 days (the duration of the 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 cm² 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.

No empirical evidence was found to assess the sensitivity of the key characterizing species, *Palmaria palmata*, to this pressure. Irvine (1983), observed morphological adaptation of the plant in fairly sheltered, silty conditions; sometimes the blade divisions are wedge-shaped and finely dissected above, or the blade has numerous linear divisions throughout. It is likely that this form reduces possible smothering that may result from increased siltation. However, this adaptation occurs in areas with chronic, relatively low levels of siltation and the blade form is unlikely to confer different levels of resistance to a single deposition event at the pressure benchmark. The perennial holdfasts may survive short periods of siltation, even though fronds may be damaged or removed aiding recovery.

Dendrodoa grossularia is a small ascidian, capable of reaching a size of approx 8.5 mm (Millar, 1954) and is, therefore, likely to be inundated by deposition of 30 cm of sediment. If inundation is long lasting then the understory community may be adversely affected.

Sensitivity assessment. Overall, the biotope is unlikely to be affected by smothering, unless adjacent to sediment. An increase in sediment could result in scour following re-suspension due to wave and tidal flow. Removal of the sediment is likely to be rapid. Overall, resistance is assessed as '**Medium**', mainly due to the scour effect as the sediment is removed, although the risk that some parts of the biotope that exist on flat or gently sloping areas may experience burial. Resilience is '**High**' and sensitivity is '**Low**'.

Litter

Not Assessed (NA)

Q: NR A: NR C: NR

Not assessed (NA)

Q: NR A: NR C: NR

Not assessed (NA)

Q: NR A: NR C: NR

Not assessed.

Electromagnetic changes

No evidence (NEv)

Q: NR A: NR C: NR

Not relevant (NR)

Q: NR A: NR C: NR

No evidence (NEv)

Q: NR A: NR C: NR

'No evidence' was found.

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

Whilst no evidence could be found for the effect of noise or vibrations on the characterizing species of these biotopes, it is unlikely that these species have the facility for detecting or noise vibrations.

Sensitivity assessment. The characterizing sponges are unlikely to respond to noise or vibrations and resistance is therefore assessed as '**High**', resilience as '**High**' and Sensitivity as '**Not sensitive**'.

Introduction of light or shading

Medium

Q: Low A: NR C: NR

High

Q: High A: Medium C: Medium

Low

Q: Low A: Low C: Low

These biotopes are characteristic of shaded or dark, moist caves, presumably because green algae would out-compete the dominant species at higher light levels. If artificial lighting was introduced to a cave where this biotope occurred, then it might adversely affect the biotopes by promoting green algal growth. Increased shade might reduce light levels below those needed for the algae to survive. While increase or decrease in light could affect the algae present within this biotope, growth ceases for a number of red algae (including *Chondrus crispus*) below ca $1.0 \mu\text{mol m}^{-2}\text{s}^{-1}$ (ca 50 Lux), whereas this value is $2 \mu\text{mol m}^{-2}\text{s}^{-1}$ (ca 100 Lux) for green algae (Leukart & Lüning, 1994)

Jones *et al.* (2012) compiled a report on the monitoring of sponges around Skomer Island and found that many sponges, particularly encrusting species, preferred vertical or shaded bedrock to open, light surfaces. However, it is possible that this relates to decreased competition with algae. Whilst no evidence could be found for the effect of light on the other faunal species present within these biotopes, it is unlikely that they would be impacted.

Sensitivity assessment. An increase at the benchmark level (0.1 Lux) is unlikely to be significant. However, shading may result in loss of the red algae and, in severe cases, the biotope may become SpR.Den or SpByAs. Resistance to this pressure is assessed as '**Medium**' and resilience as '**High**'. This biotope is therefore considered to be '**Low**' at the benchmark level.

Barrier to species movement

High

Q: Low A: NR C: NR

High

Q: High A: High C: High

Not sensitive

Q: Low A: Low C: Low

Barriers that reduce the degree of tidal excursion may alter larval supply to suitable habitats from source populations. Conversely 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 and red algal turfs 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

The pressure definition is not directly applicable to caves or overhangs, so '**Not relevant**' has been recorded. Collision via ship groundings or terrestrial vehicles is possible but the effects are probably similar to those of abrasion above.

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

'Not relevant'

Biological Pressures

Resistance

Resilience

Sensitivity

Genetic modification & translocation of indigenous species

No evidence (NEv)

Q: NR A: NR C: NR

Not relevant (NR)

Q: NR A: NR C: NR

No evidence (NEv)

Q: NR A: NR C: NR

'No evidence' was found.

Introduction or spread of invasive non-indigenous species

No evidence (NEv)

Q: NR A: NR C: NR

Not relevant (NR)

Q: NR A: NR C: NR

No evidence (NEv)

Q: NR A: NR C: NR

Scour in this biotope will probably limit the 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 are considered to also inhibit invasive algal species. The Australasian barnacle *Austrominius* (syn *Elminius*) *modestus* was introduced to British

waters on ships during the second World War. The species does well in estuaries and bays, where it can displace *Semibalanus balanoides*. However, on exposed shores, the native species out-compete this invasive species (Raffaelli & Hawkins, 1999). Two non-native spirobids – *Dexiospira oshoroensis* and *Pileolaria rose pigment ATA* - 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,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, 2012b).

Sensitivity assessment. Overall, there is ‘No evidence’ of this biotope having been adversely affected by non-native species. It should be noted that replacement of native barnacles and spirobids by non-natives alters the identity of the species present but would have little impact on biotope character and function. Due to the constant risk of new invasive species, the literature for this pressure should be revisited.

Introduction of microbial pathogens Not relevant (NR)
Q: NR A: NR C: NR

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

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

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 tropics (Littler *et al.*, 2007). No evidence was found that these pathogens are impacting temperate coralline habitats.

Gochfeld *et al.* (2012) found that diseased sponges hosted significantly different bacterial assemblages compared to healthy sponges, with diseased sponges also exhibiting a significant decline in sponge mass and protein content. Sponge disease epidemics can have serious long-term effects on sponge populations, especially in long-lived, slow-growing species (Webster, 2007). Numerous sponge populations have been brought to the brink of extinction, including cases in the Caribbean (with 70-95% disappearance of sponge specimens) (Galstoff, 1942) and the Mediterranean (Vacelet, 1994; Gaino *et al.*, 1992). Decaying patches and white bacterial film were reported in *Haliclona oculata* and *Halichondria panicea* in North Wales, 1988-89, however, no disease has been detected since (Webster, 2007). Specimens of *Cliona* spp. exhibited blackened damage since 2013 in Skomer. Preliminary results have shown that clean, fouled and blackened *Cliona* all have very different bacterial communities. The blackened *Cliona* are effectively dead and have a bacterial community similar to marine sediments. The fouled *Cliona* have a very distinct bacterial community that may suggest a specific pathogen caused the effect (Burton, pers comm; Preston & Burton, 2015). No record of diseases in the characterizing hydroids could be found. There appears to be little research into ascidian diseases, particularly in the Atlantic. The parasite *Lankesteria ascidiae* targets the digestive tubes and can cause ‘long faeces syndrome’ in *Ciona*

intestinalis (although it has also been recorded in other species). Mortality occurs in severely affected individuals within about a week following first symptoms. (Mita *et al.*, 2012).

Sensitivity assessment

Whilst evidence for disease in corallines, ascidians and sponges exists. There is little evidence of mortality in the characterizing species in the British Isles. However, given evidence of limited mortality of *Halichondria panicea*, resistance is assessed as 'Medium', resilience as 'High' and sensitivity as 'Low'.

Removal of target species

Medium

Q: Medium A: Low C: Medium

High

Q: High A: Medium C: Medium

Low

Q: Medium A: Low C: Medium

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. *Palmaria palmata* (known as dulse) is harvested from the wild for human consumption both commercially and recreationally. As the important characterizing and structuring species, extensive removal of *Palmaria palmata* could alter the character of the biotope. Garbary *et al.* (2012) studied harvested and non-harvested shores in Nova Scotia, Canada, containing stands of *Palmaria palmata*. They also conducted the experimental removal of *Palmaria palmata* and assessed simulated removal of *Palmaria palmata* by an experienced commercial harvester. Simulated commercial harvesting reduced cover of *Palmaria palmata* from 70% to 40%, although experimental removal on shores not usually harvested reduced cover to 20% (Garbary *et al.*, 2012).

It should be noted that *Mytilus edulis* and whelks may be found in this biotope and whilst abundance is likely to be low, harvesting cannot be ruled out. As these species are not characterizing, they are not considered for this pressure.

Sensitivity assessment

Based on removal of *Palmaria palmata*, resistance is assessed as 'Medium'. Resilience is assessed as 'High' (based on removal by cutting, leaving the holdfast to regenerate) and biotope sensitivity is assessed as 'Low'.

Removal of non-target species

Low

Q: Low A: NR C: NR

High

Q: High A: Medium C: Medium

Low

Q: Low A: Low C: Low

The characteristic species probably compete for space within the biotope, so that loss of one species would probably have little if any effect on the other members of the community. However, removal of the characterizing species due to trampling or similar pressures is likely to remove a proportion of the biotope. These direct, physical impacts are assessed through the abrasion pressures. The sensitivity assessment for this pressure considers any biological/ecological effects resulting from the removal of non-target species on this biotope.

Sensitivity assessment

Therefore, incidental removal of the characterizing species could result in a decline of the biotope. Resistance is assessed as 'Low', resilience 'High' and sensitivity is 'Low'.

Bibliography

- Ackers, R.G., 1983. Some local and national distributions of sponges. *Porcupine Newsletter*, **2** (7).
- Ackers, R.G.A., Moss, D. & Picton, B.E. 1992. *Sponges of the British Isles (Sponges: V): a colour guide and working document*. Ross-on-Wye: Marine Conservation Society.
- Airoidi, L., 2003. The effects of sedimentation on rocky coast assemblages. *Oceanography and Marine Biology: An Annual Review*, **41**, 161-236
- Aneiros, F., Rubal, M., Troncoso, J.S. & Bañón, R., 2015. Subtidal benthic megafauna in a productive and highly urbanised semi-enclosed bay (Ría de Vigo, NW Iberian Peninsula). *Continental Shelf Research*, **110**, 16-24.
- 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.
- Bak, U.G., 2014. Suitable cultivation areas for *Palmaria palmata* (Rhodophyta) and *Saccharina latissima* (Phaeophyceae) in the inner Danish waters in relation to variations in light, temperature and salinity. Roskilde University. MSc thesis
- Balata, D., Piazzoli, 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.
- Bamber, R.N. & Irving, P.W., 1993. The *Corallina* run-offs of Bridgewater Bay. *Porcupine Newsletter*, **5**, 190-197.
- 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. & 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., 1953. The effect of lowered salinity on some barnacle nauplii. *Journal of Animal Ecology*, **22**, 328-330.
- 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.
- Barthel, D., 1986. On the ecophysiology of the sponge *Halichondria panicea* in Kiel Bight. I. Substrate specificity, growth and reproduction. *Marine Ecology Progress Series*, **32**, 291-298.
- Barthel, D. & Wolfrath, B., 1989. Tissue sloughing in the sponge *Halichondria panicea*: a fouling organism prevents being fouled. *Oecologia*, **78**, 357-360.
- Bauvais, C., Zirah, S., Piette, L., Chaspoul, F., Domart-Coulon, I., Chapon, V., Gallice, P., Rebuffat, S., Pérez, T. & Bourguet-Kondracki, M.-L., 2015. Sponging up metals: bacteria associated with the marine sponge *Spongia officinalis*. *Marine Environmental Research*, **104**, 20-30.
- Bell, J.J. & Barnes, D.K., 2000. The distribution and prevalence of sponges in relation to environmental gradients within a temperate sea lough: inclined cliff surfaces. *Diversity and Distributions*, **6** (6), 305-323.
- Bell, J.J. & Smith, D., 2004. Ecology of sponge assemblages (Porifera) in the Wakatobi region, south-east Sulawesi, Indonesia: richness and abundance. *Journal of the Marine Biological Association of the UK*, **84** (3), 581-591.
- Bennell, S.J., 1981. Some observations on the littoral barnacle populations of North Wales. *Marine Environmental Research*, **5**, 227-240.
- Berman, J., Burton, M., Gibbs, R., Lock, K., Newman, P., Jones, J. & Bell, J., 2013. Testing the suitability of a morphological monitoring approach for identifying temporal variability in a temperate sponge assemblage. *Journal for Nature Conservation*, **21** (3), 173-182.
- 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 *Ascopyllum nodosum* and associated assemblages affected by human trampling in Portugal. *Marine Ecology Progress Series*, **439**, 73-82.
- 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: <http://www.nonnativespecies.org>
- 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.
- 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.
- Bricker, S.B., Clement, C.G., Pirhalla, D.E., Orlando, S.P. & Farrow, D.R., 1999. National estuarine eutrophication assessment: effects of nutrient enrichment in the nation's estuaries. NOAA, National Ocean Service, Special Projects Office and the National Centers for Coastal Ocean Science, Silver Spring, MD, 71 pp.
- Bricker, S.B., Longstaff, B., Dennison, W., Jones, A., Boicourt, K., Wicks, C. & Woerner, J., 2008. Effects of nutrient enrichment in the nation's estuaries: a decade of change. *Harmful Algae*, **8** (1), 21-32.
- Brosnan, D.M. & Crumrine, L.L., 1994. Effects of human trampling on marine rocky shore communities. *Journal of Experimental Marine Biology and Ecology*, **177**, 79-97.
- 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.
- Carver, C., Mallet, A. & Vercaemer, B., 2006. Biological synopsis of the solitary tunicate *Ciona intestinalis*. Canadian Manuscript Report of Fisheries and Aquatic Science, No. 2746, v + 55 p. Bedford Institute of Oceanography, Dartmouth, Nova Scotia.
- 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
- Connell, J.H., 1961. Effects of competition, predation by *Thais lapillus*, and other factors on natural populations of the barnacle *Balanus balanoides*. *Ecological Monographs*, **31**, 61-104.
- 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/>
- Crisp, D. & Patel, B., 1969. Environmental control of the breeding of three boreo-arctic cirripedes. *Marine Biology*, **2** (3), 283-295.
- 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.
- 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.
- Davison, I.R., 1991. Environmental effects on algal photosynthesis: temperature. *Journal of Phycology*, **27** (1), 2-8.
- De Vos, L., Rützler K., Boury-Esnault, N., Donadey C., Vacelet, J., 1991. *Atlas of Sponge Morphology. Atlas de Morphologie des Éponges*. Washington, Smithsonian Institution Press.
- Dixon, D.R., 1985. Cytogenetic procedures. *Pomatoceros triqueter*: A test system for environmental mutagenesis. In *The effects of stress and pollution in marine animals*.
- Dommasnes, A., 1969. On the fauna of *Corallina officinalis* L. in western Norway. *Sarsia*, **38**, 71-86.
- 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.
- Fell, P.E. & Lewandrowski, K.B., 1981. Population dynamics of the estuarine sponge, *Halichondria* sp., within a New England eelgrass community. *Journal of Experimental Marine Biology and Ecology*, **55** (1), 49-63.
- Fish, J.D. & Fish, S., 1996. *A student's guide to the seashore*. Cambridge: Cambridge University Press.
- 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.
- Fowler, S. & Laffoley, D., 1993. Stability in Mediterranean-Atlantic sessile epifaunal communities at the northern limits of their range. *Journal of Experimental Marine Biology and Ecology*, **172** (1), 109-127.
- Frazer, A.W.J., Brown, M.T. & Bannister, P., 1988. The frost resistance of some littoral and sub-littoral algae from southern New Zealand. *Botanica Marina*, **31**, 461-464.
- Fu, W., Wu, Y., Sun, L. & Zhang, W., 2007. Efficient bioremediation of total organic carbon (TOC) in integrated aquaculture system by marine sponge *Hymeniacidon perlevis*. *Biotechnology and bioengineering*, **97** (6), 1387-1397.
- Gaino, E., Frine, C. & Giuseppe, C., 2010. Reproduction of the Intertidal Sponge *Hymeniacidon perlevis* (Montagu) Along a

- Bathymetric Gradient. *Open Marine Biology Journal*, **4**, 47-56.
- Gaino, E., Pronzato, R., Corriero, G. & Buffa, P., 1992. Mortality of commercial sponges: incidence in two Mediterranean areas. *Italian Journal of Zoology*, **59** (1), 79-85.
- Galstoff, P., 1942. Wasting disease causing mortality of sponges in the West Indies and Gulf of Mexico. *Proceedings 8th American Scientific Congress*, pp. 411-421.
- Garbary, D.J., Beveridge, L.F., Flynn, A.D. & White, K.L., 2012. Population ecology of *Palmaria palmata* (Palmariales, Rhodophyta) from harvested and non-harvested shores on Digby Neck, Nova Scotia, Canada. *Algae*, **27** (1), 33-42.
- Gerrodette, T. & Flechsig, A., 1979. Sediment-induced reduction in the pumping rate of the tropical sponge *Verongia lacunosa*. *Marine Biology*, **55** (2), 103-110.
- Gittenberger, A. & Van Loon, W.M.G.M., 2011. Common Marine Macrozoobenthos Species in the Netherlands, their Characteristics and Sensitivities to Environmental Pressures. GiMaRIS report no 2011.08. DOI: [10.13140/RG.2.1.3135.7521](https://doi.org/10.13140/RG.2.1.3135.7521)
- Gochfeld, D., Easson, C., Freeman, C., Thacker, R. & Olson, J., 2012. Disease and nutrient enrichment as potential stressors on the Caribbean sponge *Aplysina cauliformis* and its bacterial symbionts. *Marine Ecology Progress Series*, **456**, 101-111.
- Gorzula, S., 1977. A study of growth in the brittle-star *Ophiocomina nigra*. *Western Naturalist*, **6**, 13-33.
- Grandy, N., 1984. *The effects of oil and dispersants on subtidal red algae*. Ph.D. Thesis. University of Liverpool.
- Guiry, M.D. & Guiry, G.M. 2015. AlgaeBase [Online], National University of Ireland, Galway [cited 30/6/2015]. Available from: <http://www.algaebase.org/>
- Hawkins, S.J. & Harkin, E., 1985. Preliminary canopy removal experiments in algal dominated communities low on the shore and in the shallow subtidal on the Isle of Man. *Botanica Marina*, **28**, 223-30.
- Hawkins, S.J. & Hartnoll, R.G., 1985. Factors determining the upper limits of intertidal canopy-forming algae. *Marine Ecology Progress Series*, **20**, 265-271.
- Hayward, P.J. & Ryland, J.S. 1994. *The marine fauna of the British Isles and north-west Europe. Volume 1. Introduction and Protozoans to Arthropods*. Oxford: Clarendon Press.
- Hayward, P.J. & Ryland, J.S. (ed.) 1995b. *Handbook of the marine fauna of North-West Europe*. Oxford: Oxford University Press.
- Hills, J. & Thomason, J., 1998. The effect of scales of surface roughness on the settlement of barnacle (*Semibalanus balanoides*) cyprids. *Biofouling*, **12** (1-3), 57-69.
- 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.
- 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.
- Huthnance, J., 2010. Ocean Processes Feeder Report. London, *DEFRA on behalf of the United Kingdom Marine Monitoring and Assessment Strategy (UKMMAS) Community*.
- Hyslop, B.T., Davies, M.S., Arthur, W., Gazey, N.J. & Holroyd, S., 1997. Effects of colliery waste on littoral communities in north-east England. *Environmental Pollution*, **96** (3), 383-400.
- Ignatiades, L. & Becacos-Kontos, T., 1970. Ecology of fouling organisms in a polluted area. *Nature* **225**, 293 - 294
- Irvine, L.M., 1983. *Seaweeds of the British Isles vol. 1. Rhodophyta Part 2A. Cryptonemiales (sensu stricto), Palmariales, Rhodymeniales*. London: British Museum (Natural History).
- Jakola, K.J. & Gulliksen, B., 1987. Benthic communities and their physical environment to urban pollution from the city of Tromsø, 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., 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.
- JNCC, 2015. The Marine Habitat Classification for Britain and Ireland Version 15.03. (20/05/2015). Available from <https://mhc.jncc.gov.uk/>
- Johansen, W.H., 1974. Articulated coralline algae. *Oceanography and Marine Biology: an Annual Review*, **12**, 77-127.
- Johnston, E.L. & Roberts, D.A., 2009. Contaminants reduce the richness and evenness of marine communities: a review and meta-analysis. *Environmental Pollution*, **157** (6), 1745-1752.
- Jones, J., Bunker, F., Newman, P., Burton, M., Lock, K., 2012. Sponge Diversity of Skomer Marine Nature Reserve. *CCW Regional Report*, CCW/WW/12/3.
- Kain, J.M., & Norton, T.A., 1990. Marine Ecology. In *Biology of the Red Algae*, (ed. K.M. Cole & Sheath, R.G.). Cambridge: Cambridge

University Press.

- Karsten, U., Dummermuth, A., Hoyer, K. & Wiencke, C., 2003. Interactive effects of ultraviolet radiation and salinity on the ecophysiology of two Arctic red algae from shallow waters. *Polar Biology*, **26** (4), 249-258.
- 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.
- Kinne, O. (ed.), 1972. *Marine Ecology: A Comprehensive, Integrated Treatise on Life in Oceans and Coastal Waters*, Vol.1, *Environmental Factors*, part 3. New York: John Wiley & Sons.
- Kinne, O., 1971b. Salinity - invertebrates. In *Marine Ecology: A Comprehensive, Integrated Treatise on Life in Oceans and Coastal Waters*. Vol. 1 *Environmental Factors*, Part 2, pp. 821-995. London: John Wiley & Sons.
- 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.
- Lancaster, J. (ed), McCallum, S., A.C., L., Taylor, E., A., C. & Pomfret, J., 2014. Development of Detailed Ecological Guidance to Support the Application of the Scottish MPA Selection Guidelines in Scotland's seas. *Scottish Natural Heritage Commissioned Report No.491 (29245)*, Scottish Natural Heritage, Inverness, 40 pp.
- Latham, H., 2008. Temperature stress-induced bleaching of the coralline alga *Corallina officinalis*: a role for the enzyme bromoperoxidase. *Bioscience Horizons*, 1-10
- Lemoine, N., Buell, N., Hill, A. & Hill, M., 2007. Assessing the utility of sponge microbial symbiont communities as models to study global climate change: a case study with *Halichondria bowerbanki*. *Porifera research: biodiversity, innovation, and sustainability. Série livros*, **28**, 239-246.
- 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.
- Leukart, P. & Lüning, K., 1994. Minimum spectral light requirements and maximum light levels for long-term germling growth of several red algae from different water depths and a green alga. *European Journal of Phycology*, **29** (2), 103-112.
- Littler, M. & Littler, D., 1998. An undescribed fungal pathogen of reef-forming crustose coralline algae discovered in American Samoa. *Coral Reefs*, **17** (2), 144-144.
- Littler, M. M., & Littler, D. S. 1984. Relationships between macroalgal functional form groups and substrata stability in a subtropical rocky-intertidal system. *Journal of Experimental Marine Biology and Ecology*, **74**(1), 13-34.
- 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
- Lohrmann, N.L., Logan, B.A. & Johnson, A.S., 2004. Seasonal acclimatization of antioxidants and photosynthesis in *Chondrus crispus* and *Mastocarpus stellatus*, two co-occurring red algae with differing stress tolerances. *The Biological Bulletin*, **207** (3), 225-232.
- 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.
- Lüning, K., 1990. *Seaweeds: their environment, biogeography, and ecophysiology*: John Wiley & Sons.
- Marques, D., Almeida, M., Xavier, J. & Humanes, M., 2007. Biomarkers in marine sponges: acetylcholinesterase in the sponge *Cliona celata*. *Porifera Research: Biodiversity, Innovation and Sustainability. Série Livros*, **28**, 427-432.
- 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.
- Millar, R.H., 1954. The annual growth and reproductive cycle of the ascidian *Dendrodoa grossularia* (van Beneden). *Journal of the Marine Biological Association of the United Kingdom*, **33**, 33-48.
- Mills, D.B., Ward, L.M., Jones, C., Sweeten, B., Forth, M., Treusch, A.H. & Canfield, D.E., 2014. Oxygen requirements of the earliest animals. *Proceedings of the National Academy of Sciences*, **111** (11), 4168-4172.
- Minchinton, T.E., Schiebling, R.E. & Hunt, H.L., 1997. Recovery of an intertidal assemblage following a rare occurrence of scouring by sea ice in Nova Scotia, Canada. *Botanica Marina*, **40**, 139-148.
- Mita, K., Kawai, N., Rueckert, S. & Sasakura, Y., 2012. Large-scale infection of the ascidian *Ciona intestinalis* by the gregarine *Lankesteria ascidia* in an inland culture system. *Diseases of aquatic organisms*, **101** (3), 185-195.
- Morgan, K.C., Shacklock, P.F. & Simpson, F.J., 1980. Some aspects of the culture of *Palmaria palmata* in greenhouse tanks. *Botanica Marina*, **23**, 765-770.
- Naylor, P., 2011. *Great British Marine Animals, 3rd Edition*. Plymouth. Sound Diving Publications.

- NBN, 2015. National Biodiversity Network 2015(20/05/2015). <https://data.nbn.org.uk/>
- Newell, R.C., 1979. *Biology of intertidal animals*. Faversham: Marine Ecological Surveys Ltd.
- Norton, T.A., 1992. Dispersal by macroalgae. *British Phycological Journal*, **27**, 293-301.
- O'Brien, P.J. & Dixon, P.S., 1976. Effects of oils and oil components on algae: a review. *British Phycological Journal*, **11**, 115-142.
- 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.
- Prendergast, G.S., Zurn, C.M., Bers, A.V., Head, R.M., Hansson, L.J. & Thomason, J.C., 2009. The relative magnitude of the effects of biological and physical settlement cues for cypris larvae of the acorn barnacle, *Semibalanus balanoides* L. *Biofouling*, **25** (1), 35-44.
- Preston J. & Burton, M., 2015. Marine microbial assemblages associated with diseased Porifera in Skomer Marine Nature Reserve (SMNR), Wales. Aquatic Biodiversity and Ecosystems, 30th August – 4th September, Liverpool., pp. p110.
- 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.
- Riisgård, H.U., Bondo Christensen, P., Olesen, N.J., Petersen, J.K, Moller, M.M. & Anderson, P., 1993. Biological structure in a shallow cove (Kertinge Nor, Denmark) - control by benthic nutrient fluxes and suspension-feeding ascidians and jellyfish. *Ophelia*, **41**, 329-344.
- 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>
- Robbins, J.V., 1978. Effects of physical and chemical factors on photosynthetic and respiratory rates of *Palmaria palmata* (Florideophyceae), In *Proceedings of the ninth International Seaweed Symposium, Santa Barbara, California, USA, 20-27 August 1977*, (ed. Jensen, A. & Stein, J.R.), 273-283. Science Press, Princeton, NJ, USA.
- 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.
- Różycki, O. & Gruszzyński, M., 1991. On the infauna of an Arctic estuary Nottinghambukta, Svalbard. *Polish Polar Research*, **12** (3), 433-444.
- Ryland, J.S. & De Putron, S., 1998. An appraisal of the effects of the *Sea Empress* oil spillage on sensitive invertebrate communities. *Countryside Council for Wales Sea Empress Contract Report*, no. 285, 97pp.
- Saunders, R. & Connell, S., 2001. Interactive effects of shade and surface orientation on the recruitment of spirorbid polychaetes. *Austral Ecology*, **26** (1), 109-115.
- Schönberg, C.H.L., 2015. Happy relationships between marine sponges and sediments—a review and some observations from Australia. *Journal of the Marine Biological Association of the United Kingdom*, 1-22.
- Seapy, R.R. & Littler, M.M., 1982. Population and Species Diversity Fluctuations in a Rocky Intertidal Community Relative to Severe Aerial Exposure and Sediment Burial. *Marine Biology*, **71**, 87-96.
- 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., 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.
- Taylor, A. R. A., Chen, L. C. M., Smith, B. D., & Staples, L. S. 1981. Chondrus holdfasts in natural populations and in culture. In *Proceedings of the International Seaweed Symposium* **8**, 140-145.
- 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.

Thomassen, S. & Riisgård, H.U., 1995. Growth and energetics of the sponge *Halichondria panicea*. *Marine Ecology Progress Series*, **128**, 239-246.

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.

Tyler-Walters, H. & Arnold, C., 2008. Sensitivity of Intertidal Benthic Habitats to Impacts Caused by Access to Fishing Grounds. *Report to Cynfor 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>

Vacelet, J., 1994. Control of the severe sponge epidemic—Near East and Europe: Algeria, Cyprus, Egypt, Lebanon, Malta, Morocco, Syria, Tunisia, Turkey, Yugoslavia. *Technical Report—the struggle against the epidemic which is decimating Mediterranean sponges FI: TCP/RAB/8853*. Rome, Italy. 1–39 p, pp.

Van der Meer, J.P. & Chen, C-M., 1979. Evidence for sexual reproduction in the red algae *Palmaria palmata* and *Halosaccion ramentaceum*. *Canadian Journal of Botany*, **57**, 2452-2459.

Vethaak, A.D., Cronie, R.J.A. & van Soest, R.W.M., 1982. Ecology and distribution of two sympatric, closely related sponge species, *Halichondria panicea* (Pallas, 1766) and *H. bowerbanki* Burton, 1930 (Porifera, Demospongiae), with remarks on their speciation. *Bijdragen tot de Dierkunde*, **52**, 82-102.

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.

Webster, N.S., 2007. Sponge disease: a global threat? *Environmental Microbiology*, **9** (6), 1363-1375.

Webster, N.S. & Taylor, M.W., 2012. Marine sponges and their microbial symbionts: love and other relationships. *Environmental Microbiology*, **14** (2), 335-346.

Werner, A. & Dring, M., 2011. Cultivating *Palmaria palmata*. *Aquaculture Explained*, **27**, *Bord Iascaigh Mhara* (BIM).

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

Wethey, D.S., Woodin, S.A., Hilbish, T.J., Jones, S.J., Lima, F.P. & Brannock, P.M., 2011. Response of intertidal populations to climate: effects of extreme events versus long term change. *Journal of Experimental Marine Biology and Ecology*, **400** (1), 132-144.

Wiedemann, T., 1994. *Oekologische Untersuchungen in Gezeitentümpeln des Helgolaender Nord-Ost Felswatts*. , Diploma thesis, University of Kiel, Germany.

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>

Wulff, J., 2006. Resistance vs recovery: morphological strategies of coral reef sponges. *Functional Ecology*, **20** (4), 699-708.

Zahn, R., Zahn, G., Müller, W., Kurelec, B., Rijavec, M., Batel, R. & Given, R., 1981. Assessing consequences of marine pollution by hydrocarbons using sponges as model organisms. *Science of The Total Environment*, **20** (2), 147-169.