Himanthalia elongata and red seaweeds on exposed to moderately exposed lower eulittoral rock

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

Dr Heidi Tillin & Georgina Budd

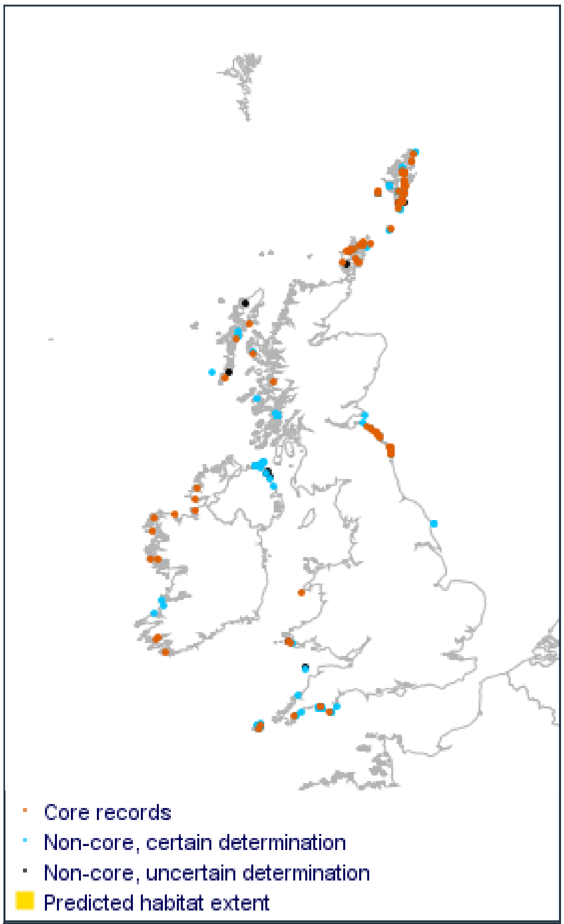
2016-02-08

A report from:
The Marine Life Information Network, Marine Biological Association of the United Kingdom.

This review can be cited as:
Tillin, H.M. & Budd, G., 2016. [Himanthalia elongata] and red seaweeds on exposed to moderately exposed lower eulittoral rock. 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.360.1

(page left blank)

Himanthalia elongata **and** ***Laminaria digitata*** **on lower shore bedrock.**
Photographer: Keith Hiscock
Copyright: Joint Nature Conservation Committee (JNCC)



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

Researched by Dr Heidi Tillin & Georgina Budd **Refereed by** This information is not refereed.

Summary

UK and Ireland classification

EUNIS 2008	A1.123	*Himanthalia elongata* and red seaweeds on exposed lower eulittoral rock
JNCC 2015	LR.HLR.FR.Him	*Himanthalia elongata* and red seaweeds on exposed to moderately exposed lower eulittoral rock
JNCC 2004	LR.HLR.FR.Him	*Himanthalia elongata* and red seaweeds on exposed to moderately exposed lower eulittoral rock
1997 Biotope	LR.ELR.FR.Him	*Himanthalia elongata* and red seaweeds on exposed lower eulittoral rock

Description

Exposed to moderately exposed lower eulittoral bedrock characterized by the wrack Himanthalia elongata with a dense turf of red seaweeds beneath. Himanthalia elongata may occur on tide-swept, sheltered shores in sea lochs (e.g. Loch Maddy). The wrack *Fucus serratus* is normally present as well. The predominant red seaweeds are usually *Mastocarpus stellatus*, *Osmundea pinnatifida*, *Corallina officinalis* and *Palmaria palmata* that tend to grow over a crust of the pink coralline algae *Lithothamnion* spp. Any patches between the algal turf may be colonized by barnacles *Semibalanus balanoides*, or *Perforatus perforatus* in the south-west, and by the limpet *Patella vulgata*. Pits and crevices in the rock often provide a refuge for the whelk *Nucella lapillus*, the winkle *Littorina* spp. and small individuals of the mussel *Mytilus edulis*. Besides the dominant seaweeds there are a number of other red, brown and green seaweeds present. These include species such as the red seaweeds *Dumontia contorta*, *Lomentaria articulata*, *Porphyra* spp., the kelp *Laminaria digitata* and the green seaweeds *Ulva intestinalis*, *Ulva lactuca* and *Cladophora rupestris*.

On some shores this biotope may occur as a distinct zone between a *Fucus serratus* and red algal turf (Fser.R) and the kelp *Alaria esculenta* and *Laminaria digitata* community (Ala.Ldig). This biotope generally characterizes those shores which are too exposed for F. serratus to form a dense canopy, often occurring as large patches within the *Fucus serratus* / red seaweed turf zone (Fser.R). Consequently, *Fucus serratus* plants frequently occur amongst the *Himanthalia elongata* and red seaweed turf.

In early spring only characteristic *Himanthalia elongata* buttons are present, while the long erect parts of the thalli appears later in the season. This biotope may therefore appear very similar to Coff.Coff biotope in the spring and care should be taken when assessing these biotopes in early spring. (See Connor *et al*., 2004; JNCC).

Depth range

-

Additional information

No text entered.

Listed By

- none -

Further information sources

Search on:

G G JNCC

Habitat review

Ecology

Ecological and functional relationships

In the lower eulittoral zone, limpet grazing, which limits the upward extension of low-shore species in to the mid-eulittoral (e.g. Underwood, 1979; Underwood, 1980; Underwood & Jernakoff, 1981), becomes less effective and algal turfs of *Himanthalia elongata*, red algae and kelp species dominate the lower shore. Physical conditions in the lower eulittoral may be optimal for macroalgae. Several species of fucoids grow faster lower on the shore than higher on the shore (Schonbeck & Norton, 1980). Rapid growth to attain a size greater than 5 cm, grants immunity from limpet grazing and enhances survival (see Hawkins & Hartnoll, 1983 for review).

Brown algal canopy:

- *Fucus serratus* cannot withstand too much turbulence created by wave action, whilst *Fucus serratus* persists in the biotope, under wave exposed conditions the biotope is dominated by *Himanthalia elongata*. The cover provided by the 'straps' of *Himanthalia elongata* varies throughout the year (see recruitment processes) but the canopy it provides probably enhances the survival of an undergrowth of red algae by providing protection from excessive insolation, desiccation or by inhibiting the growth of potential rivals.
- *Himanthalia elongata* supports a considerable epifauna and epiflora (see epiphytic species below) described by Kitching (1987).

Red algal turf and coralline algal crust:

- Red algae represent a major source of food for a variety of herbivores, including molluscs, crustaceans and sea urchins. Grazers may truncate the vertical distribution of some red algae or eliminate all but the most tolerant grazer-resistant species, but *Chondrus crispus*, *Osmundea pinnatifida*, and many calcareous crustose species in the biotope seem in some circumstances to be reliant upon browsing fauna to remove epiphytes or other competitors that might otherwise smother the plant (Kain & Norton, 1990).
- Species such as *Lomentaria articulata*, *Plumaria elegans* and *Cladophora rupestris* prefer the shade of rocky overhangs or a screen of larger fucoids, and may become scarce if shaded surfaces are lacking. *Osmundea pinnatifida* can become more abundant further upshore in the eulittoral zone as long as it is shaded. *Corallina officinalis*, however, can rise further upshore in open sunlight than in shade provided that it is in damp depressions or runnels, indicating that strong sunlight does not harm it when desiccation is minimal (Lewis, 1964).

Epiphytic species:

- To obtain a better position in the 'struggle for light', smaller, non-parasitic algal species may grow as epiphytic algae on larger plants, and in most instances are only superficially attached to their host (Lüning, 1990). Epiphytic algae decrease the growth rate of their host, increase the probability of breakage, and may reduce reproductive output (d' Antonio, 1985).
- On exposed shores the 'straps' of *Himanthalia elongata* have few epiphytes, however in slightly more sheltered locations they may develop a 'microforest' of filamentous algae, which provides a habitat for a variety of faunal epiphytes. Kitching (1987) recorded 105 species of flora and fauna on *Himanthalia elongata* plants in Lough Hyne. The upper surface

of the buttons were very resistant to colonization by epiphytes while the underside of the buttons were usually colonized by Bryozoa and spirorbid worms. The straps were frequently covered in a microforest of filamentous algae, such as *Ectocarpus siliculosus*, *Ceramium pedicellatum* and *Ulva prolifera*. Fauna such as the gastropods *Rissoa pavida*, juvenile *Patella pellucida* and amphipods, were often associated with the filamentous epiphytes. The cushion-like brown algae *Elachista scutulata* occurs exclusively on *Himanthalia elongata* (Lüning, 1990). As a protection against epiphytes, young actively growing plants of *Himanthalia elongata* regularly shed off the outer layer of their outer-most cell wall (Filion-Myklebust & Norton, 1981). However, *Elachista scutulata* by-passes the cell shedding mechanism if its host by growing in cryptostomata (invaginations with hairs) and other natural breaks in the host thallus surface, where cells are not removed (Russell & Veltkamp, 1984).

- Cell-shedding has also been detected in crustose coralline algae that grow beneath the brown algal canopy. In this instance, however, it is a case of ridding their surfaces of algal gametophytes (Masaki *et al*., 1981).
- Where competition for space and light restricts the occurrence of *Palmaria palmata* on rock, the species often has an epiphytic habit on other algae, especially kelps.

Sessile and mobile epilithic species:

- Species such as the barnacles, *Semibalanus balanoides* and *Chthamalus* spp., limpet, *Patella vulgata*, gastropod molluscs, *Littorina neglecta* and *Nucella lapillus*, and the mussel, *Mytilus edulis*, are dominant species within the biotope, although as algal cover increases, so does competition for space.
- Barnacles remain most numerous in the most wave exposed situations in the biotope, especially on vertical surfaces, steep slopes facing seawards and mounds rising above seaweed covered flats as recumbent fronds of macroalgae and red algal turf make flat surfaces less hospitable for barnacles. Algal competition for space in the lower eulittoral not only curtails the distribution of *Semibalanus balanoides* and *Chthamalus* spp. but also suppresses the penetration of *Perforatus perforatus* up the shore (Lewis, 1964).
- Density and size of *Patella vulgata* in the lower eulittoral is highly variable, and although a common species in the ELR.Him biotope, numbers in macroalgae dominated biotopes are usually less than on barnacle dominated biotopes but individuals can achieve a greater size (Lewis, 1964).
- *Nucella lapillus* is an important intertidal predator and preys mainly on barnacles and mussels but may also prey on cockles, other bivalves and gastropods. Crothers (1985) suggested a mean annual consumption of 15-20 mussels per dog whelk (Largen, 1967a; Bayne & Scullard, 1978) and reported rates of 0.5 or 0.59 mussels/day or 1.1 *Semibalanus balanoides* /day in summer (Connell, 1961; Fretter & Graham, 1962; Anala, 1974). Dog whelks avoid dense mussel beds, preferring the diffuse margins between the mussel bed and the surrounding barnacle dominated substratum, or solitary mussels (Petraitis, 1987; Fretter & Graham, 1994; Davenport *et al*., 1996). This was partly because mussels can immobilise gastropods (*Nucella lapillus* and *Littorina* spp.) by entangling the gastropods with their byssus threads.
- Many other animals are found in the biotope, but do not make a distinct contribution to the appearance of the shore. However, a species that may be apparent is the anemone *Actinia equina* that benefits from the damp conditions of pits and crevices amongst algae. Numerous small and specialised habitats exist within the biotope. On the algae or on damp rock beneath gammarid amphipods, the isopod *Idotea*, small hydroids and bryozoans (*Dynamena pumila* and *Flustrellida hispida* especially) and other small gastropod molluscs

may be found.

Seasonal and longer term change

Species within the biotope demonstrate seasonality in terms of growth and reproduction. For instance, germlings of *Himanthalia elongata* become visible on the shore in early March and form buttons with an average size of 10-25 mm by August. Those buttons which grow to 15 mm by November produce receptacles that autumn. The receptacles grow little in length during autumn and winter but increase rapidly between February and May and may reach 2 m in length. *Chondrus crispus* is a perennial species whose holdfasts may persist on the substratum for several years but whose growth is annual, following die-back. It is fertile throughout the winter and can therefore take advantage of surfaces that become available for colonization, after other species have died back (Kain, 1975). Storms and increased wave action are more likely to occur in the winter months and may cause physical damage to the community. Dudgeon & Johnson (1992) noted wave induced disturbance of intertidal *Chondrus crispus* on shores of the Gulf of Maine, USA, during winter. 25-30 % of cover of large *Chondrus crispus* thalli was lost in one winter. Physical disruption of the algal turf is likely to promote diversity as spaces become available for colonization.

Habitat structure and complexity

Bedrock forms the substratum of the biotope, the pits, crevices and inclination of which create microhabitats exploitable by both mobile and sessile epilithic species. In addition the algal species of the community add considerable structural complexity to the biotope in the form of additional substratum for settlement by epiphytic species. The straps of *Himanthalia elongata* form a canopy over the substratum, the cover of which varies throughout the year according to the stage of growth. The straps of *Himanthalia elongata* grow from February onwards so that by the summer they shade the understorey vegetation and create a local microclimate while the tide is out.

Productivity

Rocky shore communities are highly productive and are an important source of food and nutrients for members of neighbouring terrestrial and marine ecosystems (Hill *et al*., 1998). Macroalgae exude considerable amounts of dissolved organic carbon which are taken up readily by bacteria and may even be taken up directly by some larger invertebrates. Only about 10 % of the primary production is directly cropped by herbivores (Raffaelli & Hawkins, 1996). Dissolved organic carbon, algal fragments and microbial film organisms are continually removed by the sea. This may enter the food chain of local, subtidal ecosystems, or be exported further offshore. Rocky shores make a contribution to the food of many marine species through the production of planktonic larvae and propagules which contribute to pelagic food chains. For instance, the mean release per month from each square meter of a stand of *Chondrus crispus* was estimated to be 961 X 10^6 carpospores and 204 X 10^6 tetraspores (Bhattacharya, 1985). Measurements of the productivity of benthic algae are relatively few, particularly for the Rhodophyta (Dixon, 1973). Blinks (1955) estimated the net production of red algae to be in the order of 11 to 54 g dry weight per m▯ per day.

Recruitment processes

- *Himanthalia elongata* has a life history and growth pattern unique among the Fucales. The species invests 98 % of its total biomass in reproductive rather than vegetative tissue. It is usually has a biennial lifecycle, reproducing once and then dying. The reproductive bodies

or receptacles take the form of long straps, which sprout from the centre of the button. Gametes are released from June until winter. Usually germlings become visible on the shore in early March and form buttons with an average size of 10-25 mm by August. Those buttons which grow to 15 mm by November produce receptacles that autumn. The receptacles grow little in length during autumn and winter but increase rapidly between February and May. From June onwards, adult plants release gametes on a low tide by liberating them into mucus, which dribbles onto the substratum below.
- All the spores of red algae are non flagellate and dispersal is wholly a passive process (Fletcher & Callow, 1992). Spores do however vary in their sinking rate as determined by size and density. In general, due to the difficulties of re-entering the benthic boundary layer, it is likely that successful colonization is achieved under conditions of limited dispersal and/or minimum water current activity. Norton (1992) reported that although spores may travel long distances (e.g. *Ulva* sp. 35 km, *Phycodrys rubens* 5 km), the reach of the furthest propagule does not equal useful dispersal range, and most successful recruitment probably occurs within 10 m of the parent plants. It is expected, therefore, that recruitment of, for example, *Chondrus crispus*, *Palmaria palmata* and the majority of other macroalgae in the biotope would occur from local populations and that establishment and recovery of isolated populations would be patchy and sporadic. Scrosati *et al.* (1994) commented that viability of spores of *Chondrus crispus* was low (<30 %) and suggested that reproduction by spores probably does not contribute much to maintenance of the intertidal population in Nova Scotia, compared to vegetative growth of gametophytes.
- Vadas *et al.* (1992) reviewed recruitment and mortality of early post settlement stages of benthic algae. They identified 6 intrinsic and 17 extrinsic factors affecting recruitment and mortality. They concluded that grazing, canopy and turf effects were the most important but that desiccation and water movement may be as important for the early stages. The review indicated that recruitment is highly variable and episodic and that mortality of algae at this period is high. Chance events during the early post settlement stages are therefore likely to play a large part in survival.
- See full *MarLIN* reviews for specific information on reproduction and longevity: *Chondrus crispus*, *Palmaria palmata*, *Corallina officinalis*, *Fucus serratus*.
- Many animal species in the biotope, possess a planktonic larval stage which float in the plankton before settling and metamorphosing into adult form, for example; *Mytilus edulis*, *Patella vulgata* and *Semibalanus balanoides*. This strategy allows species to rapidly colonize new areas that become available such as in the gaps on otherwise algal dominated rocks created by storms. For these organisms it has long been evident that recruitment from the pelagic phase is important in governing the density of populations on the shore (Little & Kitching, 1996). Both the demographic structure of populations and the composition of assemblages may be profoundly affected by variation in recruitment rates. *Nucella lapillus* is oviparous, it lays its fertilized eggs directly on the substratum.

Time for community to reach maturity

From the information available on recruitment of important characterizing species, it is likely that the community would reach maturity within five years.

Additional information

No text entered

Preferences & Distribution

Habitat preferences

Depth Range	
Water clarity preferences	
Limiting Nutrients	Nitrogen (nitrates)
Salinity preferences	Full (30-40 psu)
Physiographic preferences	Open coast
Biological zone preferences	Lower eulittoral
Substratum/habitat preferences	Bedrock
Tidal strength preferences	
Wave exposure preferences	Exposed, Moderately exposed
Other preferences	

Additional Information

Nitrogen is the primary resource that limits seaweed growth and consequently variations in seaweed growth should parallel variations in nitrogen supply (Lobban & Harrison, 1997).

Species composition

Species found especially in this biotope

- *Himanthalia elongata*
- *Patella vulgata*
- *Semibalanus balanoides*

Rare or scarce species associated with this biotope

-

Additional information

No text entered

Sensitivity review

Sensitivity characteristics of the habitat and relevant characteristic species

This biotope is characterized by the wrack *Himanthalia elongata* with a dense turf of red seaweeds beneath. The wrack *Fucus serratus* is normally present as well. The predominant red seaweeds are usually *Mastocarpus stellatus*, *Osmundea pinnatifida*, *Corallina officinalis* and *Palmaria palmata* that tend to grow over a crust of the pink coralline algae *Lithothamnion* spp. The algal species within this biotope are considered to define the character of the biotope and to provide the habitat that supports other associated species and the sensitivity assessments are therefore based on these species. Any patches between the algal turf may be colonized by barnacles *Semibalanus balanoides* or *Perforatus perforatus* in the south-west and by the limpet *Patella vulgata*. Pits and crevices in the rock often provide a refuge for the whelk *Nucella lapillus*, the winkle *Littorina* spp. and small individuals of the mussel *Mytilus edulis*. The sensitivity of associated green and red species and limpets and barnacles are described very generally as these are not considered key characterizing, structuring or functional species within this biotope. More information on these species can be found in the assessments for other biotopes on this website.

Resilience and recovery rates of habitat

Himanthalia elongata has a life history and growth pattern unique among the Fucales, typically the species has a biennial life cycle, reproducing once and then dying. *Himanthalia elongata* has a two stage morphology. A small button-like frond is first produced, from which large strap-like reproductive fronds are formed. The button stage is clubbed shaped at first and then develops into a button shape 2-3 cm in diameter, which is connected to the substrate by a holdfast and short stipe. Each button typically produces two strap-like reproductive fronds in autumn, although plants have been observed with one to four straps. Usually, germlings become visible on the shore in early March and form buttons with an average size of 10-25 mm by August. Those buttons which grow to 15 mm by November produce receptacles that autumn. The receptacles grow little in length during autumn and winter but increase rapidly between February and May. When the plants are fertile the straps become mottled with brown spots, each spot with a pale centre marking the opening to the conceptacle. The species invests 98 % of the total biomass in reproductive rather than vegetative tissue (Brenchley *et al*., 1996). Time of reproduction is strongly site dependent, probably due to water temperature. Population and age-structure studies suggest the plant is biennial although some plants grow vegetatively for 2 years before reproducing in the third year (Russell, 1990).

Zygotes of the species are very large in comparison to most seaweeds. They are spherical, comparatively heavy and measure 0.2mm across so that they rapidly settle to the substratum (Moss *et al.* 1973). After fertilization, there is a long period of 5-7 days before attaching rhizoids develop. During this period the zygote is anchored to the substratum using the fertilization membrane, which is expanded into a wide brim (Moss *et al*., 1973). Zygotes are incapable of growing on silt, but germlings are tolerant of temporary cover by drifting sand. Gamete dispersal is thought to be limited so recruitment from external populations is probably low. Early germling growth is probably strongly influenced by the presence of adults, or other foliose algae providing protection from desiccation, wave action and high irradiances, although shading probably limits growth rates of the germlings (Stengel *et al*., 1999; Stengel pers. comm., previous MarLIN review).

The species recruited to concrete blocks placed in the intertidal zone (within an existent population of *Himanthalia elongata* at Finavarra West Ireland) at an average level of 45 buttons per block (968 sq. cm in area) in March of the first year, dropping to only 4 or 5 buttons per block by early summer, but rising to 1500 buttons per block by March of the second year (Stengel *et al*., 1999). The timing of the placement of blocks, when most gametes had been released, most likely contributed to the low recruitment observed in the first year (Stengel et al., 1999). Survival and growth of buttons in West Ireland were higher in red algal turfs and lower on exposed rock so that the availability of suitable micro-habitats influences distribution (Stengel *et al*., 1999). Where the canopy is entirely cleared recruitment and recovery may be prolonged. At Artedeo, (northern Spain) plots 30 cm^2 were cleared (using a paint scraper and wire brush) at different tidal heights (Viejo, 2009). At a low tidal level where *Himanthalia elongata* dominated with *Fucus vesiculosus* present, cleared plots recovered more slowly than those higher in the intertidal. The canopy forming species had recovered 4 years after disturbance but differences in understorey algae persisted (Viejo, 2009). Recovery from complete removal, even at small scales, is therefore assessed as ‘Medium’.

Turf forming red algae may recover through repair and regrowth of damaged fronds or via recolonization of rock surfaces where all the plant material is removed. Red algae (Rhodophyta) exhibit distinct morphological stages over the reproductive life history, a phenomenon is known as heteromorphy. The thalli can regrow from basal crusts or holdfasts where they remain supporting the recovery of the biotope (Mathieson & Burns, 1975; Dudgeon & Johnson, 1992). The basal crusts of *Corallina officinalis* and the petrocelis (tetrasporophyte) stage of *Mastocarpus stellatus* are perennial, tough, resistant stages that prevent other species from occupying the rock surface and allow rapid regeneration and are therefore a significant recovery mechanism. *Palmaria palmata* is a perennial species with new growth every year from the small discoid holdfast which could remain for several years. Where fronds are damaged regrowth will be rapid during the spring-summer growth season and older parts of individuals typically have small 'leaflets' along the margin especially where damaged. The holdfasts of individual *Chondrus crispus* coalesce over time and can form an extensive crust on rock (Taylor *et al*., 1981). The crust of *Mastocarpus stellatus* is known as the Petrocelis stage and can also cover extensive areas. Littler & Kauker (1984) suggest that the crustose bases of *Corallina officinalis* were adapted to resist grazing and desiccation whereas the fronds were adapted for higher primary productivity and reproduction. The basal crusts are tougher than the upright fronds (requiring a pressure of 94 g/mm^2 to penetrate compared to 43 g/mm^2). Regeneration of the basal crusts provides a more rapid route to recovery than recolonization. Experiments in the intertidal in southern California found that areas scraped back to crusts recovered four times more rapidly than sterilised plots where the crusts were removed (Littler & Kauker, 1984). Turfs of *Osmundea pinnatifida* also expand through lateral, vegetative growth (Prathep, 2001).

The spores of red algae are non-motile (Norton, 1992) and therefore entirely reliant on the hydrographic regime for dispersal. 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 red algae would normally only recruit from local populations and that recovery of remote populations would be much more protracted. In areas of bare rock on the Isle of Man monitored for a year on shores that were suitable for turfs no recruitment of *Osmundea pinnatifida* was observed and vegetative growth rather than recolonization is probably a key factor (Prathep, 2001). Small gaps within the red algal turf may provide a suitable microhabitat for recolonization with recolonization of *Osmundea pinnatifida* outside of turfs depending on favourable conditions and the presence of microhabitats such as crevices providing shelter and excluding grazers (Prathep, 2001).

Most of the evidence for recovery of *Chondrus crispus* is based on experiments that simulate the effects of different harvesting mechanisms and intensities (Macfarlane, 1952; Mathieson & Burns, 1975). Due to similarities in biology, it is suggested that these recovery rates may also be applicable to *Mastocarpus stellatus*. Macfarlane (1952) in a series of experiments identified that where *Chondrus crispus* was removed by cutting of fronds or thorough raking (leaving the crusts undamaged) the turf had recovered and there were no notable differences between the experimental areas and control sites. However, where the crusts were removed by scraping or damaged the experimental plots were still recovering nearly two years after the treatment. Following experimental harvesting by drag raking (holdfasts and small blades undamaged) in New Hampshire, USA, populations recovered to 1/3 of their original biomass after 6 months and totally recovered after 12 months (Mathieson & Burns, 1975). The authors suggested that control levels of biomass and reproductive capacity are probably re-established after 18 months of regrowth (where crusts are not removed). It was noted, however, that time to recovery was much extended if harvesting occurred in the winter, rather than the spring or summer (Mathieson & Burns, 1975). Where the bases are removed then recolonization will require more time. 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 2 years, *Chondrus crispus* had re-established approximately 50% cover on the lower shore and after 5 years it was the dominant macroalga at this height, with approximately 100% cover. The authors pointed out that although *Chondrus crispus* was a poor colonizer, it was the best competitor. Pringle and Semple (1980) estimated it would take about four years for a bare patch in a *Chondrus* bed to fill in with harvestable plants and five to ten years for *Chondrus* to re-establish in barren areas.

Areas that are cleared during the reproductive period have the potential to be rapidly colonized. *Corallina officinalis* was shown to settle on artificial substances within 1 week of their placement in the intertidal in New England summer (Harlin & Lindbergh, 1977). However, settlement plates laid out in the autumn were not recolonized until the next spring. In the lower rocky intertidal in southern California dominated by *Corallina officinalis* with foliose overstorey algae present, Littler & Kauker (1984) experimentally cleared plots and followed the recovery for 12 months. Some areas were scraped allowing the basal crusts to remain whereas others were completely sterilised (removal of all material and surfaces then scorched with a blow torch to remove bases). In scraped plots, up to 15% cover of *Corallina officinalis* fronds returned within three months after removal of fronds and all other epiflora/fauna (Littler & Kauker, 1984) while in sterilized plots (all basal crusts removed) appearance of articulated fronds occurred 6 months following clearance. At the end of the 12 month observation period, *Corallina officinalis* cover had increased to approximately 18% in plots where basal crusts remained and to approximately 10% in sterilised plots. Similarly, Bamber & Irving (1993) reported that new plants grew back in scraped transects within 12 months, although the resistant crustose bases were probably not removed.

A number of papers by Edyvean & Ford (1984a & b; 1986;1987) describe aspects of reproduction and growth of encrusting coralline, *Lithophyllum incrustans*. Studies by Edyvean & Forde (1987) in populations of *Lithophyllum incrustans* in Pembroke south-west Wales suggest that reproduction occurs on average early in the third year. Reproduction may be sexual or asexual. Populations release spores throughout the year but abundance varies seasonally, with the populations studied in Cullercoats Bay, and Lannacombe Bay (North East and South West England, respectively) producing fewer spores in the summer. Spore release is initiated by changes in temperature or salinity (see relevant pressure information) at low tide so that spore dispersal is restricted to within the tide pool enhancing local recruitment. Spore survival is extremely low with only a tiny proportion of spores eventually recruiting to the adult population (Edyvean & Ford, 1986). The spores are released from structures on the surface called conceptacles, these are formed annually and subsequently buried by the new layer of growth. Plants can be aged by counting the number of layers of conceptacles. Edyvean & Ford (1984) found that the age structure of populations sampled from Orkney (Scotland) Berwick (northern England) and Devon (England) were similar, mortality seemed highest in younger year classes with surviving individuals after the age of 10 years appear relatively long-lived (up to 30 years). In St Mary's Northumberland, the population was dominated by the age 6-7 year classes (Edyvean & Ford, 1984). Growth rates were highest in young plants measured at Pembroke (south-west Wales) with an approximate increase in diameter of plants of 24 mm in year class 0 and 155 mm in year 1 and slowing towards an annual average horizontal growth rate of 3mm/year (Edyvean & Ford, 1987).

Some repair of damaged encrusting coralline occurs through vegetative growth. Chamberlain (1996) observed that although *Lithophyllum incrustans* was quickly affected by oil during the Sea Empress spill, recovery occurred within about a year. The oil was found to have destroyed about one third of the thallus thickness but regeneration occurred from thallus filaments below the damaged area. Recolonization by propagules is an important recovery mechanism, Airoldi (2000) observed that encrusting coralline algae recruited rapidly on to experimentally cleared subtidal rock surfaces in the Mediterranean Sea, reaching up to 68% cover in 2 months. As encrusting corallines are sensitive to desiccation (Dethier, 1994) it should be noted that these subtidal habitats are probably more favourable for recruitment, growth and survival than intertidal rock pools.

Turf forming biotopes and fucoid dominated biotopes may represent alternate stable states that continue while the dominant turf or fucoids occupy space. Removal of the turf may, therefore, allow re-establishment of a Fucoid or kelp dominated biotope that will remain until environmental or other factors again alter the state. Lubchenco (1980) for example, on shores in New England, found that the removal of *Chondrus crispus* turf allowed the establishment of *Fucus* spp. Removal of grazers and the turf allowed *Fucus* spp. to establish 100% cover, highlighting the significance of grazers in structuring the biotope. Exclusion of grazing limpets, on shores in southern Britain (Swanage and Heybrook), led to the colonization of red algal turfs by *Himanthalia elongata* and *Fucus serratus* within 2 years (Boaventura *et al*., 2002). MacFarlane (1952) also recorded a shift to a *Corallina officinalis* and encrusting coralline biotope following over raking (for harvesting) of *Chondrus crispus* turf, in these areas gastropods had increased in abundance and prevented the recovery of *Chondrus crispus* by grazing. It should, therefore, be noted that where red algal turfs are removed, recovery may be prolonged.

Resilience assessment. Recruitment and survival of *Himanthalia elongata* are enhanced by the presence of adults due to low dispersal rates and the requirement for shelter. Resilience is therefore assessed as 'High' (within 2 years) where resistance is 'High' or 'Medium' (where only sub-lethal effects or the removal of <25% of the adults within the biotope is predicted). Where the biotope is more extensively impacted and resistance is 'Low' or 'None', recovery of the *Himanthalia elongata* canopy is considered to be 'Medium' (2-10 years). Recovery may occur more rapidly where red algal turfs are present as these enhance recruitment (Stengel *et al*., 1999; Boaventura *et al*., 2002). Recovery rates of the red algae will be greatly influenced by whether the crust stages remain from which the thalli can regrow. Where the bases remain and resistance is assessed as 'Medium' (loss of <25% of individuals or cover) then recovery is assessed as 'High' based on regrowth from crusts and remaining plants. Where resistance is assessed as 'Low' or 'None' and a high proportion of bases are lost then recovery may be more protracted. Based on recovery of *Chondrus crispus* from ice scour (Minchington *et al*., (1997) and experiments by Littler & Kauker, (1984) recovery is assessed as 'Medium' (2-10 years). Recruitment of the associated invertebrates, barnacles, limpets and littorinids, is considered to follow the same trajectories as the algae, as populations may recover rapidly where resistance is 'Medium' but require longer due to episodic success where resistance is 'Low' or 'None'. Where turfs are removed over large areas, recovery will depend on the supply of propagules from neighbouring populations and as dispersal is limited, the recovery will depend on the supply of propagules which will be influenced site-specific factors, particularly local water transport. Caveats regarding possible state shifts where beds are extensively removed should also be considered when applying sensitivity assessments. Identifying tipping points for shifts to alternate stable states is problematic, therefore although the recovery rates based on examples and life history traits are used in the assessments these may underestimate recovery time which will be influenced by pressure and site-specific factors. The removal of *Himanthalia elongata* from this biotope would probably lead to reclassification to the similar biotope LR.HLR.FR.Coff.Coff.

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.

Hydrological Pressures

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

Species found in the intertidal 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 and species that occur higher on the shore are generally able to tolerate a greater range of temperatures than those confined to lower shore levels (Davenport & Davenport, 2005), although the timing of site-specific factors such as low tides will influence local acclimation. For intertidal species increased temperatures may also result in desiccation when exposed (see changes in emergence pressure). The key characterizing species *Himanthalia elongata* has a cold-temperate distribution, its southern distribution limit is northern Spain (Martínez *et al*., 2015) and in that area its range has been contracting, possibly due to increased temperatures (Duarte *et al*., 2013). An examination of survey records from 1974- 2010 correlated with summer and winter surface temperatures (Yesson *et al*., 2015) suggest that in the UK *Himanthalia elongata* is negatively correlated with winter temperatures (e.g. abundance is greater where winters are colder), the abundance of *Himanthalia elongata* could therefore decline due to chronic or acute increases in winter temperatures.

Experiments have been conducted to determine the physiological threshold tolerances to temperature of *Himanthalia elongata* collected from shores at Redondela (Ria de Vigo, Spain) and Moledo (northern Portugal). Fronds were kept in tanks and maintained at one of a range (from

8.3 °C to 29.8 °C) of 12 temperatures for 6 weeks with 20 replicates for each temperature. Mortality and frond growth (weight to nearest 0.1g) were measured weekly (Martínez *et al*., 2015). A marked growth decline occurred at 18°C, this temperature was also the threshold where the probability of survival was 0.5, e.g. 50 % of plants at this temperature were likely to die. These findings were supported by field observation, as the species was absent from locations with average August sea surface temperatures higher than 19 °C (Martínez *et al*., 2015).

Increased temperatures may affect reproduction and survival of early stages. Himanthalia elongata plants commenced gamete release on the west coast of Ireland in June at two sites where water temperatures were 16-18 °C (Stengel *et al*., 1999). Gamete release may be triggered by temperature or by photoperiod as the release also coincides with longer day length. Moss *et al*., (1973) tested the effects of different levels of light intensity and temperature on the growth of early stages of *Himanthalia elongata* These laboratory experiments indicated that gamete release could occur at a range of temperatures between 10 and 20 °C.. Zygotes kept at 22 °C and 25 °C failed to germinate at all light intensities (total darkness to 6000 Lux).

The red algae associated with this biotope occur at higher shore levels and many have broad geographic distributions, repeated increases in temperature at the pressure benchmark may lead to replacement of sensitive species by those that are more tolerant. *Palmaria palmata* is largely found in northern temperate and arctic waters. With the species southern limits in Portugal in Europe and northern Spain (Garbary *et al*., 2012 and references therein). In the laboratory, plants only became fertile if left at temperatures between 5-7°C with a short light period (Meer van der, 1979). 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).

Mastocarpus stellatus may be found higher on the shore and its distribution suggests that it has a higher tolerance to increased temperatures and desiccation than *Chondrus crispus*. *Chondrus* is found extensively throughout Europe and North America records also recorded under a number of synonyms from Africa and Asia (Guiry & Guiry, 2015). Spore germination in *Chondrus crispus* appears to be temperature dependent with spores discharged at temperatures of 5 °C failing to germinate although in laboratory culture at 10 °C spores were viable all year round (Bhattacharya, 1985). In New Hampshire, USA, *Chondrus crispus* grows abundantly in waters with an annual variation in surface temperature from -1 to +19°C (Mathieson & Burns, 1975). Prince & Kingsbury (1973) reported cessation of growth in *Chondrus crispus* cultures at 26 °C, first mortality of spores at 21.1 °C and total mortality of spores at 35-40°C, even if exposed for just 1 minute.

The calcareous red algae *Corallina officinalis* has a cosmopolitan distribution (Guiry & Guiry, 2015) and throughout its range experiences wide variation in temperatures (although local populations may be acclimated to the prevailing thermal regime). Littler & Kauker (1984) suggested that the crustose bases of *Corallina officinalis* are more resistant of desiccation or heating than fronds. Severe damage was noted in *Corallina officinalis* fronds as a result of desiccation during unusually hot and sunny weather in summer 1983. An abrupt increase in temperature of 10 °C caused by the hot, dry 'Santa Anna' winds (between January and February) in Santa Cruz, California resulted in die back of several species of algae exposed at low tide (Seapy & Littler, 1982). Lüning (1990) reported that *Corallina officinalis* from Helgoland survived one week exposure to temperatures between 0 °C and 28 °C. 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 a 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. Samples kept at 5, 15, 20 and 25 °C 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.

Coralline crusts, including *Lithophyllum incrustans* are found further south than the UK and are considered to tolerate increased temperatures (although they may be more sensitive to drying rather than higher temperatures). 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 tide pools with a smaller volume of water that were more exposed to temperature and salinity changes due to lower buffering capacity. Sexual plants (or the spores that give rise to them) were suggested to be more susceptible than asexual plants to extremes of local environmental variables (temperature, salinity etc.) as they occur with greater frequency at sites where temperature and salinity were more stable (Edyvean & Ford, 1984b). *Lithophyllum incrustans* is close to the northern edge of its range and is likely to tolerate increased temperatures.

In an exceptionally hot summer (1983, with an increase of between 4.8 and 8.5 °C) Hawkins & Hartnoll (1985) observed no temperature bleaching of adult *Himanthalia elongata* (although some buttons were bleached) or other canopy forming species. However, understorey red algae showed more signs of damage with bleached *Corallina officinalis* and 'lithothamnia' observed around the edges of pools due to dessication. Occasional damaged specimens of *Palmaria palmata*, *Osmundea pinnatifida* and *Mastocarpus stellatus* were observed. This evidence suggests that this biotope would tolerate a chronic or acute increase in temperature at the pressure benchmark.

Most of the other species within the biotope are distributed to the north and south of Britain and Ireland and unlikely to be adversely affected by a chronic temperature change. Sandison (1968); (reported unpublished in Lewis, 1964) noted that heat coma occurred in *Nucella lapillus* at 27 -28 °C, and death at 32 -33 °C. Newell (1979) noted that oxygen consumption (hence metabolic rate) fell with decreased temperature and starvation, being low in winter but high in summer. This resulted in a high scope for activity, and dog whelks responded rapidly to increases in temperature in the spring. Newell (1979) pointed out that dog whelks could adjust their metabolic rate with temperature and season. Stickle *et al.* (1985) also noted that feeding and ingestion rates decreased with decreasing temperature and salinity. Increased temperatures increase the risk of desiccation, especially on sheltered shores. However, dog whelks demonstrate behavioural adaptations depending on the type of shore they inhabit, e.g. dog whelks from sheltered shores forage less in sunny, warm weather, whereas animals from wave exposed shores (higher humidity) favoured calm periods even when sunny (Burrows & Hughes, 1989; Fretter & Graham, 1994). Crothers (1985) suggested that the southern limit of dog whelk distribution was temperature dependant and noted that in Portugal dog whelks live inside mussels clumps and in Massachusetts, where water temperature may reach 25 °C, dog whelks may spend summer below the tide mark. Therefore, *Nucella lapillus* is probably relatively tolerant of temperature change within the normal range for the UK, and is probably tolerant to a change of 2 °C over a year. However, an acute temperature change (e.g. 5 °C) will probably interfere with feeding activity and in summer may result in direct mortality or indirect mortality due to heat coma and desiccation. Limpets, *Patella vulgata* and littorinids occur within this biotope. Laboratory studies suggest that adults of these species can tolerate temperature increases. The median upper lethal temperature limit in laboratory tests on *Littorina littorea*, *Littorina saxatilis* was approximately 35 ^{oC} (Davenport &

Davenport, 2005). *Patella vulgata* can also tolerate high temperatures. The body temperature of *Patella vulgata* can exceed 36 °C in the field, (Davies, 1970); adults become non-responsive at 37-38 °C and die at temperatures of 42 °C (Evans, 1948).

Sensitivity assessment. The key characterizing species *Himanthalia elongata* and some of the associated species such as *Palmaria palmata* and *Patella vulgata* are close to the southern limit of their distribution. Adults are considered to be able to withstand an acute and chronic increase in temperature at the pressure benchmark, therefore biotope resistance is assessed as 'High' and resilience as 'High' and the biotope is considered to be 'Not sensitive'. It should be noted that the timing of acute and chronic increases would alter the degree of impact and hence sensitivity. An acute change occurring on the hottest days of the year and exceeding thermal tolerances would lead to mortality. The available evidence indicates that adults have greater temperature tolerances than juveniles, germlings or spores. A chronic increase in temperature (for longer than the pressure benchmark) may therefore lead to recruitment failures in this biotope of *Himanthalia elongata*, *Patella vulgata*, *Semibalanus balanoides* and other species close to the southern limit of distribution.

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

The key characterizing species *Himanthalia elongata* has a cold-temperate distribution, its southern distribution limit is northern Spain (Martínez *et al*., 2015) and in that area its range has been contracting, possibly due to increased temperatures (Duarte *et al*., 2013). An examination of survey records from 1974-2010 correlated with summer and winter surface temperatures suggest that in the UK *Himanthalia elongata* is negatively correlated with winter temperatures (e.g. abundance is greater where winters are colder), suggesting that in the UK lower temperatures are beneficial for this species (Yesson *et al*., 2015).

Experiments have been conducted to determine the physiological threshold tolerances to temperature of *Himanthalia elongata* collected from shores at Redondela (Ria de Vigo, Spain) and Moledo (northern Portugal). Fronds were kept in tanks and maintained at one of a range of 12 temperatures (8.3 °C to 29.8 °C) for 6 weeks with 20 replicates for each temperature. Mortality and frond growth (weight to nearest 0.1g) were measured weekly (Martínez *et al*., 2015). Over the course of the experiment 16 °C appeared to be the optimal temperature for growth. However, plants maintained at the lowest temperature 8 °C still grew at similar rates to those kept at 10 °C and 12 °C. No further evidence was found to suggest sensitivity to lower temperatures.

Under extremely low temperatures, components of the community demonstrate tolerance. 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). Lüning (1990) suggested that most littoral algal species were tolerant of cold and freezing. For example, the photosynthetic rate of *Chondrus crispus* recovered after 3hrs at -20 °C but not after 6 hrs (Dudgeon *et al*., 1990). The photosynthetic rate of *Mastocarpus stellatus* higher on the shore fully recovered from 24 hrs at -20 °C. 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). *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 and references therein). In the laboratory, plants only became fertile if left at temperatures between 5-7°C with a short light period (Meer van der, 1979). Temperatures at or above 15°C may induce physiological stress (Werner & Dring, 2011; Morgan *et al*., 1980)

The associated species are also likely to be tolerant of a decrease in temperature at the pressure benchmark. *Mytilus edulis* are eurytopic, found in a wide temperature range and in areas which frequently experience freezing conditions and are vulnerable to ice scour (Seed & Suchanek 1992). The tolerance of *Semibalanus balanoides* collected in the winter (and thus acclimated to lower temperatures) to low temperatures was tested in the laboratory. The median lower lethal temperature tolerance was -14.6 °C (Davenport & Davenport, 2005). A decrease in temperature at the pressure benchmark is therefore unlikely to negatively affect this species. The same series of experiments indicated that median lower lethal temperature tolerances for *Littorina saxatilis* and *Littorina littorea* were -16.4 and -13 °C respectively. Adults of *Patella vulgata* are also largely unaffected by short periods of extreme cold. Ekaratne & Crisp (1984) found adult limpets continuing to grow over winter when temperatures fell to -6 °C, and stopped only by still more severe weather. However, loss of adhesion after exposure to -13 °C has been observed with limpets falling off rocks and therefore becoming easy prey to crabs or birds (Fretter & Graham, 1994). However, in the very cold winter of 1962-3 when temperatures repeatedly fell below 0 °C over a period of 2 months large numbers of *Patella vulgata* were found dead (Crisp, 1964). Periods of frost may also kill juvenile *Patella vulgata*, resulting in recruitment failures in some years (Bowman & Lewis, 1977). In colder conditions an active migration by mobile species may occur down the shore to a zone where exposure time to the air (and hence time in freezing temperatures) is less.

Sensitivity assessment. Based on the characterizing and associated species, this biotope is considered to have 'High' resistance and 'High resilience (by default) to this pressure and is therefore considered to be 'Not sensitive'. The timing of changes and seasonal weather could result in greater impacts on species. An acute decrease in temperature coinciding with unusually low winter temperatures may exceed thermal tolerances and lead to mortalities of the associated species although this would not alter the character of the biotope.

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

Local populations may be acclimated to the prevailing salinity regime and may therefore exhibit different tolerances to other populations subject to different salinity conditions and therefore caution should be used when inferring tolerances from populations in different regions. This biotope is found in full (30-35 ppt) salinity (Connor *et al*., 2004). Heavy rainfall, followed by tidal inundation can cause dramatic fluctuations in salinity and therefore species found in the intertidal are adapted to some degree to changes in salinity. Little empirical evidence was found to assess sensitivity to this pressure. Species that occur at mid to high shore levels or in rockpools are exposed to greater salinity changes and are therefore more likely to tolerate changes at the pressure benchmark. The key characterizing species *Himanthalia elongata* is confined to the lower shore and would therefore be predicted to have lower tolerance to hypersalinity than other species in the biotope that are also found at higher shore levels.

Wright & Reed (1985) exposed *Himanthalia elongata* samples to hypersaline media (51 ppt, 68 ppt or 102 ppt) for 3 days. The plants were kept at 15 ^{o}C in seawater for less than 48 hours prior to the experiment. Compared to control plants maintained at natural salinities (34 ppt) photosynthesis of plants maintained at 51 ppt decreased by approximately 20% compared with control plants after 1 day and 42% after 3 days. Greater changes occurred at more extreme salinities: plants maintained for 3 days at 102 ppt photosynthesised as <20% of the rate of control plants (Wright & Reed, 1985). *Himanthalia elongata* may tolerate a short-term increase (for perhaps a week or more) but this will affect photosynthesis and over longer time periods this is likely to lead to mortality.

The associated algal turf species are typically found in a range of salinities. 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). *Chondrus crispus* is found in a range of salinities across its range and has been reported from sites with yearly salinity range 0-10 psu and 10-35 psu (Lindgren & Åberg 1996) and sites from an average of 26-32 psu. However, at different salinities the ratio between the abundance of the tetrasporophyte phase and the gametophyte alters (Guidone & Grace, 2010). Mathieson & Burns (1971) recorded maximum photosynthesis of *Chondrus crispus* in culture at 24 psu, but rates were comparable at 8, 16 and 32 psu. Photosynthesis continued up to 60 psu. Bird *et al.* (1979) recorded growth of Canadian *Chondrus crispus* in culture between 10 and 50 psu, with a maximum at 30 psu. *Chondrus crispus* would therefore appear to be euryhaline and tolerant of a range of salinities. *Corallina officinalis* is found in rockpools, such as this biotope, where salinities may fluctuate markedly during exposure to the air. Kinne (1971) cites 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).

Young *Littorina littorea* inhabit rockpools where salinity may increase above 35 psu. Thus, key species may be able to tolerate some increase in salinity. *Mytilus edulis* is found in a wide range of salinities from variable salinity areas (18-35 ppt) such as estuaries and intertidal areas, to areas of more constant salinity (30-35 ppt) in the sublittoral (Connor *et al*., 2004). Furthermore, mussels in rockpools are likely to experience hypersaline conditions on hot days, Newell (1979) recorded salinities as high as 42 psu in intertidal rock pools, suggesting that *Mytilus edulis* can tolerate high salinities.Kirby *et al.* (1994b) simulated the effects of hyper-osmotic shock due to evaporation of mantle cavity retained seawater in *Nucella lapillus*. No mortalities were observed over the duration of the experiment where individuals were exposed to 35, 45, 55, 65 and 75 psu over periods of 6, 12 and 24 hrs at 15 °C. Overall, it appears that *Nucella lapillus* would tolerate an acute, short-term increase in salinity, albeit at metabolic cost, suggesting an intolerance of 'low'.

Sensitivity assessment. Although some increases in salinity may be tolerated by the associated species present the natural variation, (rather than the pressure benchmark) is generally short-term and mitigated during tidal inundation. This biotope is considered, based on distribution of *Himanthalia elongata* on the lower shore, to be sensitive to a persistent increase in salinity to > 40 ppt. Resistance is therefore assessed as 'Low' and recovery as 'Medium' (following restoration of usual salinity). Sensitivity is therefore assessed as 'Medium'. The associated red algal turf and invertebrates may have a greater tolerance than *Himanthalia elongata* to this pressure but some changes in abundance and composition are likely to occur where changes are long-term.

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

This biotope is recorded in full salinity habitats (Connor et al., 2004). However, high rainfall will reduce salinity on rocksurfaces when exposed to air and may create a surface layer of brackish/nearly freshwater for a period. Heavy rainfall, followed by tidal inundation can cause dramatic fluctuations in salinity. Local populations may be acclimated to the prevailing salinity regime and may therefore exhibit different tolerances to other populations subject to different salinity conditions and therefore caution should be used when inferring tolerances from populations in different regions. This biotope is found in full (30-35 ppt) salinity (Connor *et al*., 2004) and therefore a change at the pressure benchmark relates to a decrease in salinity to reduced (18-30 ppt) or variable (18-35 ppt). Species that occur at mid to high shore levels or in rockpools are exposed to greater salinity changes and are therefore more likely to tolerate changes at the pressure benchmark. The key characterizing species *Himanthalia elongata* is confined to the lower shore and would therefore be predicted to have lower tolerance to hyposalinity than other species in the biotope that are also found at higher shore levels.

Wright & Reed (1985) exposed *Himanthalia elongata* samples to hyposaline media (2 ppt, 8 ppt or 17 ppt) for 3 days. The plants were kept at 15 °C in seawater for less than 48 hours prior to the experiment. Compared to control plants maintained at natural salinities (34 ppt) photosynthesis of plants maintained at 17 ppt decreased by approximately 30% compared with control plants after 1 day and 50% after 3 days. Greater changes occurred at more extreme salinities: plants maintained for 3 days at 2 ppt photosynthesised at <30% of the rate of control plants (Wright & Reed, 1985). *Himanthalia elongata* may tolerate a short-term increase (for perhaps a week or more) but this will affect photosynthesis and over longer time periods this is likely to lead to mortality. The *Himanthalia elongata* maintained in extreme hyposaline medium (2 and 8 ‰ for longer than 1 day developed bladder like structures around the thallus (Wright & Reed, 1985). Experiments in laboratory culture have also shown that the fertilized eggs of *Himanthalia elongata* are very sensitive to even a slight drop in salinity. The percentage which germinates starts to fall as soon as the salinity falls and by the time the salinity is at 50% of normal seawater (value not given but assumed to be approximately 17 ppt) no germination takes place. Low tolerance for reductions in salinity may explain the absence of this species from river estuaries and from rocks adjacent to them (Moss *et al*., 1973).

The associated algal turf species are typically found in a range of salinities. *Chondrus crispus* is found in a range of salinities across its range and has been reported from sites with yearly salinity range 0-10 psu and 10-35 psu (Lindgren & Åberg 1996) and sites from an average of 26-32 psu. However, at different salinities the ratio between the abundance of the tetrasporophyte phase and the gametophyte alters (Guidone & Grace, 2010). Mathieson & Burns (1971) recorded maximum photosynthesis of *Chondrus crispus* in culture at 24 psu, but rates were comparable at 8, 16 and 32 psu. Photosynthesis continued up to 60 PSU. Bird *et al.* (1979) recorded growth of

Canadian *Chondrus crispus* in culture between 10 and 50 psu, with a maximum at 30 psu. *Chondrus crispus* would therefore appear to be euryhaline and tolerant of a range of salinities. 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). *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).

Corallina officinalis is found in rockpools, such as this biotope, where salinities may fluctuate markedly during exposure to the air. In the Baltic, *Corallina officinalis* is confined to deeper waters as surface salinity decreases (Kinne, 1971a), suggesting that full salinity is required in the long-term although short-term fluctuations may be tolerated (although the thresholds of this tolerance are not clear). Kinne (1971a) cites maximal growth rates for *Corallina officinalis* between 33 and 38 psu in Texan lagoons, so that a decrease in salinity at the pressure benchmark would be predicted to lead to reduced growth.

Edyvean & Ford (1984b) report 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).

Based on occurrence in estuaries it is clear that some of the species associated with this biotope have a high tolerance for this pressure. *Littorina littorea* is found in waters of full, variable and reduced salinities (Connor *et al*., 2004) and so populations are considered tolerant of decreases in salinity at the pressure benchmark. *Mytilus edulis* is found in a wide range of salinities from variable salinity areas (18-35ppt) such as estuaries and intertidal areas, to areas of more constant salinity (Connor *et al*., 2004). *Mytilus edulis* was recorded to grow in a dwarf form in the Baltic sea where the average salinity was 6.5 psu (Riisgård *et al*., 1993). In the long-term (weeks) *Mytilus edulis* can acclimate to lower salinities (Almada-Villela, 1984; Seed & Suchanek, 1992; Holt *et al*.,1998). Almada-Villela (1984) reported that the growth rate of individuals exposed to only 13 psu reduced to almost zero but had recovered to over 80% of control animals within one month. In extreme low salinities, e.g. resulting from storm runoff, large numbers of mussels may be killed (Keith Hiscock pers comm). However, Bailey *et al*. (1996) observed very few mortalities when exposing *Mytilus edulis* to a range of salinities as low as 0 ppt for two weeks at a range of temperatures. It was also noted that there was a fast recovery rate.

Sensitivity assessment. Although some daily changes in salinity may be experienced these will be mitigated during tidal inundation. This biotope is considered, based on *Himanthalia elongata*, *Corallina officinalis* and *Lithophyllum incrustans* to be sensitive to a decrease in salinity at the pressure benchmark. Resistance is therefore assessed as 'Low' and recovery as 'Medium' (following restoration of usual salinity). Sensitivity is therefore assessed as 'Medium'.

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

No evidence to assess this pressure was found. *The key characterizing species Himanthalia elongata* can tolerate fairly strong currents, as evidenced by its presence in tidal rapids in Lough

Ine, Ireland. *Himanthalia elongata* morphology varies according to the strength of water currents, (Stengel *et al*., 1999) this response may reflect environmental factors such as enhanced nutrients and/or reflect response to stressors, e.g. streamlining to reduce drag. Longer repeatedly branched plants appear more common at sites with strong currents and less wave action (Stengel, pers comm, reported in Stengel *et al*., 1999), and may reflect lower mechanical stress on the plants where flows are unidirectional rather than the oscillatory movements associated with wave action. Some macroalgae may delay gamete release during periods of high water flow and this may effect settlement and recruitment (Coelho *et al*., 2000). However at the benchmark this is not considered to significantly affect the biotope over the course of a year.

Biogenic habitat structures reduce the effects of water flows on individuals by slowing and disrupting flow. The receptacles of *Himanthalia elongata* where these are dense will reduce the flow experienced by the canopy and by the understorey algal turf and associated species. Boller and Carrington (2006), for example, found that the canopy created by a turf of *Chondrus cripsus* reduced drag forces on individual plants by 15-65%.

The coralline crusts that occur in this biotope are securely attached and as these are flat are subject to little or no drag, the red algal turf is also likely to tolerate changes in water flows at the pressure benchmark. Biotopes dominated by turfs of *Corallina officinalis* and containing similar associated species are found in a range of flow rates from 'moderately strong' (0.5-1.5 m/s) to very 'weak' (negligible) (Connor *et al*., 2004, Dommasnes, 1969) Moderate water movement is beneficial to seaweeds as it carries a supply of nutrients and gases to the plants and removes waste products. However, if flow becomes too strong, plants may become displaced. Additionally, an increase to stronger flows may inhibit settlement of spores and remove adults or germlings. However, *Corallina officinalis* and the red algae have a compact, turf forming growth which reduce water flow through turbulence and friction and are probably resistant to displacement by an increase in water flow.

Palmaria palmata is found in a range of water flow regimes from moderately strong to weak. In increasing water flow rate the downstream deflection of the thallus caused by flowing water increases the plants rate of growth by presenting the thallus perpendicular to the incident light. Experiments have shown that the flexibility of the very morphologically similar *Grateloupia turuturu* allows the species to show high reconfiguration rates (reduction in size when aligned with flow) to reduce drag and hence the risk of damage and dislodgement (Boller & Carrington, 2007). Similarly, *Mastocarpus stellatus* and *Chondrus crispus* can flex and reconfigure to reduce the size of the alga when aligned with the direction of flow, this minimises drag and hence the risk of dislodgement (Boller & Carrington, 2007). These characteristics allow these species to persist on shores that experience a range of flow speeds. For example, *Chondrus crispus* and *Mastocarpus stellatus* occur at sites in Maine, USA experiencing peak Autumn flow speeds as measured by current meters of 9.2 m/s and 5.8 m/s. At the more exposed sites with higher flow speeds some *Chondrus crispus* were dislodged during the winter.

The associated invertebrate species are considered to be tolerant of changes in water flow at the pressure benchmark. Both laboratory and field observations indicate that flow is an important factor with effects on feeding, growth and recruitment in *Semibalanus balanoides* (Sanford *et al*., 1994, Leonard *et al*., 1998), however, the results suggest that flow is not a limiting factor determining the overall distribution of barnacles as they can adapt to a variety of flow speeds. *Patella vulgata* also inhabits a range of tidal conditions and is therefore, likely to tolerate a change in water flow rate. The streamlined profile of limpet shells is of importance in increasing their tolerance of water movement, and this is undoubtedly one factor in determining the different shape of limpets at different exposures. With increasing exposure to wave action the shell develops into a low profile reducing the risk of being swept away. The strong muscular foot and a thin film of mucus between the foot and the rock enables *Patella vulgata* to grip very strongly to the substratum (Fretter & Graham, 1994). The ability of limpets to resist accelerating water flows, as distinct from constant currents, may set a limit to the kind of habitat which they can occupy and limit the size to which they can grow.*Littorina littorea* is found in areas with water flow rates from negligible to strong, although populations exposed to different levels of flow may have adapted to local conditions. Increases in water flow rates above 6 knots may cause snails in less protected locations (e.g. not in crevices etc) to be continually displaced into unsuitable habitat so that feeding may become sub-optimal. Thus, populations of *Littorina littorea* are likely to reduce. Shell morphology within littorinids varies according to environmental conditions, in sheltered areas shell apertures are small to inhibit predation where *Carcinus maenas* is more prevalent. In exposed areas the foot surface is larger to allow greater attachment and the shell spire is lower to reduce drag (Raffaelli 1982, Crothers, 1992).

Sensitivity assessment. This biotope is considered to have 'High' resistance and 'High' resilience to this pressure at the benchmark and is therefore assessed as 'Not sensitive. Changes in water flow at the pressure benchmark may result in increased or decreased sediment deposition, these changes are not considered to alter the character of the biotope but may alter species richness of the small invertebrates associated with the turf or any sediment deposits and may lead to sub-lethal abrasion of fronds. Siltation pressures associated with the deposition of high-levels of fine sediments are assessed separately.

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

Emergence regime is a key factor structuring this (and other) intertidal biotopes. This biotope is found on the lower shore and may only be exposed at the lowest tides, in some locations the species occurs subtidally (Moss *et al*., 1973). This biotope may occur as a distinct zone between a *Fucus serratus* and red algal turf (Fser.R) and the kelp *Alaria esculenta* and *Laminaria digitata* community (Ala.Ldig) (Connor *et al*., 2004).

Himanthalia elongata may physiologically tolerate an increase in the period of emergence because it has been found to grow 2m vertically further up the shore in the absence of limpets (Southward & Southward, 1978). Moss (1969) found that when *Himanthalia elongata* have been kept out of water for some time e.g. during low tides, the oospheres shed on the surface of the receptacles are unable to escape from their mucilaginous envelope and so do not get fertilized. Gametes are therefore likely to be discharged under water (Moss,1969) and changes in emergence are likely to reduce reproductive success. Therefore, an increase in the period of emergence would probably result in a depression of the species upper limit on the shore.

The red, turf-forming algae are more likely to be tolerant of an increase in emergence than *Himanthalia elongata*, as many of the species present in this biotope are also found in the mid-shore although it should be noted that the protection of canopy forming species may be a key factor determining tolerance. A study by Lubchenco (1980) on the coast of New England suggested that desiccation is an important factor determining the upper limit of *Chondrus crispus*. The species was found to extend into the mid-intertidal where it was found underneath a fucoid canopy. Removal of this canopy resulted in bleached, dried out and dead plants within two to three weeks (Lubchenco, 1980). Hawkins & Hartnoll (1985) also observed bleaching of red algae species including *Chondus crispus* and *Mastocarpus* (as *Gigartina*) *stellata*, during an unusually hot summer; the loss of canopy forming algae exacerbated the impact. Occurrence of encrusting coralline algae seems to be critically determined by exposure to air and sunlight. Colonies of encrusting coralline algae survive in the mid-shore in damp conditions, under algal canopies or in pools, but not on open rock where desiccation effects are important. Spore release by the crusting coralline *Lithophyllum incrustans* is triggered by small changes in salinity and temperature and therefore changes in emergency may alter patterns in reproduction and recruitment (see relevant pressures for further information). However, this species does occur both high and low in the intertidal (Edyvean & Ford, 1986) where conditions allow.

Tolerances vary between turf-forming species and increased emergence may alter the composition of the turf but not significantly alter the character of this elelemt. Studies have demonstrated that *Chondrus crispus* is more sensitive to freezing and desiccation than *Mastocarpus stellatus* (Mathieson & Burns, 1971; Dudgeon *et al*., 1989) which probably limits the height on which it grows on the shore. Replacement of *Chondrus crispus* by *Mastocarpus stellatus* may occur in response to increased emergence and desiccation, rather than temperature, may be the determining factor.

Increased emergence may reduce habitat suitability for the associated invertebrate species, although *Nucella lapillus*, *Mytilus edulis*, *Patella vulgata*, *Semibalanus balanoides* and littorinids are found at a range of shore levels and are found in abundance at higher shore levels than this biotope typically occurs at. These species are therefore considered not to be affected by increased emergence. Experimental grazer removal has allowed algae including *Palmaria palmata*, *Ceramium* sp. and *Osmundea* (as *Laurenci*a) *pinnatifida* to grow higher on the shore (during winter and damp summers) than usual suggesting that grazing rather than physiological tolerances may also limit the upper shore extent of this biotope. An increase in grazers and grazing within this biotope associated with reduced predation level by crabs and predators following increased emergence may remove algae but grazing on the characterizing, mature red algae turf is probably limited and may serve to check the growth of green algae. Decreased emergence is likely to increase predation by crabs and starfish.

Himanthalia elongata and Corallina officinalis and many of the associated species are found subtidally and can tolerate some decrease in emergence. Decreased emergence is likely to lead to the habitat the biotope is found in becoming more suitable for the lower shore species generally found below the biotope, leading to replacement by, for example, a kelp dominated biotope with red algae and *Corallina officinalis* surviving under the canopy.

Sensitivity assessment. Significant long-term changes in emergence would result in changes in environmental factors and grazing pressures and lead to increased competition from species better adapted to the changed conditions. The key characterizing species *Himanthalia elongata* is considered to be particularly sensitivie to increased emergence although the red algal turf may survive and be colonized by fucoids. Decreases in emergence are likely to result in colonization by perennial Kelps and *Alaria esculenta* that are more competitively suited to the changed conditions. Loss of *Himanthalia elongata* would lead to reclassification of this biotope. Biotope resistance to both increased and decreased emergence is therefore assessed as ‘Low’ based on *Himanthalia elongata*. Resilience is assessed as ‘Medium’ (where some plants survive in adjacent habitats and plants are able to support recolonization) and sensitivity to this pressure is assessed as ‘Medium’.

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

This biotope is recorded from locations that are judged to range from moderately exposed or exposed (Connor *et al*., 2004). The degree of wave exposure influences wave height, as in more exposed areas with a longer fetch waves would be predicted to be higher. As this biotope occurs across a range of exposures, this was therefore considered to indicate, by proxy, that biotopes in the middle of the wave exposure range would tolerate either an increase or decrease in significant wave height at the pressure benchmark.

Himanthalia elongata has been observed at both wave exposed and semi-exposed sites in West Ireland (Stengel et al., 1999). Survival and growth of *Himanthalia elongata* buttons was greater in algal turfs (as in this biotope) which was suggested to result from protection from wave action and desiccation (Stengel pers comm, cited in Stengel *et al*., 1999). The morphology of *Himanthalia elongata* plants may alter depending on the degree of wave exposure. Stengel (pers comm, reported in Stengel *et al*., 1999), found that at more wave exposed sites plants were smaller and had narrower receptacles. This presumable reduces the drag forces on the plant and reduces levels of breakage and detachment.

Dommasnes, (1969) recorded *Corallina officinalis* turfs from very wave sheltered areas in Norway. Colonies of *Lithophyllum incrustans* appear to thrive in conditions exposed to strong water movement. Irvine & Chamberlain (1994) observe that the species is best developed on wave exposed shores.

Chondrus crispus typically occurs in 'sheltered' and 'moderately exposed areas' (Dixon & Irvine, 1977). Strong wave action is likely to cause some damage to fronds of both the key characterizing species resulting in reduced photosynthesis and compromised growth. Furthermore, individuals may be damaged or dislodged by scouring from sand and gravel mobilized by increased wave action (Hiscock, 1983). Dudgeon & Johnson (1992) noted wave induced disturbance of intertidal *Chondrus crispus* on shores of the Gulf of Maine where 25-30% of cover of large *Chondrus crispus* thalli was lost in one winter. *Chondrus crispus* suffers greater losses than *Mastocarpus stellatus* in areas of increased wave action as the drag forces on the thallus are greater for the larger and bushier plants (Dudgeon & Johnson, 1992; Pratt & Johnson, 2002). Greater dislodgement of the faster growing Chondrus crispus by wave action during winter storms is a factor mediating competition between the two key characterizing species (Pratt & Johnson, 2002).

Gutierrez & Fernandez (1992) described morphological variability of *Chondrus crispus* according to wave exposure and emersion. They identified two well defined morphotypes; filiform and planiform. The filiform morphotype had fewer dichotomies per unit length, a circular cross section, narrow fronds and was abundant in the low intertidal and at more exposed sites. The planiform morphotype had more dichotomies, was smaller, with a flattened cross section, broader fronds and was abundant higher up the shore and in more sheltered areas. An increase or decrease in wave exposure greater than the pressure benchmark may therefore lead to shifts between the filiform and planiform morphyotypes.

Sensitivity assessment. The biotope is found across a range of wave exposures, mid-range populations are considered to have 'High' resistance to a change (increase or decrease) in significant wave height at the pressure benchmark. Resilience is assessed as ‘High’, by default, and the biotope is considered ‘Not sensitive’. *Himanthalia elongata* is more tolerant of wave action than *Fucus serratus*. A decrease in the level of wave action could result in *Himanthalia elongata* being displaced by faster growing fucoids such as *Fucus serratus*.

Chemical Pressures

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

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

Contamination by non-synthetic chemicals, at levels greater than the pressure benchmark may adversely impact the biotope. Uptake of heavy metals from solution by seaweed is influenced by factors such as light, algal nitrogen content, frond age, length of emersion, temperature, salinity, season of the year and presence of other pollutants in the surrounding water (see Lobban & Harrison, 1997) and consequently seaweed may not accurately reflect metal concentrations in the surrounding water. No information was found concerning the effects of heavy metals on *Himanthalia elongata*, turf forming and 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. Little information was found concerning the intolerance of *Chondrus crispus* to heavy metals. Burdin & Bird (1994) reported that both gametophyte and tetrasporophyte forms accumulated Cu, Cd, Ni, Zn, Mn and Pb when immersed in 0.5 mg/l solutions for 24 hours. No effects were reported however, and no relationship was detected between hydrocolloid characteristics and heavy metal accumulation. Cole *et al.* (1999) reported that Hg was very toxic to macrophytes. The sub-lethal effects of Hg (organic and inorganic) on the sporelings of an intertidal red algae, *Plumaria elegans*, were reported by Boney (1971). 100% growth inhibition was caused by 1 ppm Hg.

Most of the information available suggests that adult gastropod molluscs are rather tolerant of heavy-metal toxicity (Bryan, 1984). Winkles may absorb metals from the surrounding water by absorption across the gills or from their diet, and evidence from experimental studies on *Littorina littorea* suggest that diet is the most important source (Bryan *et al.*, 1983). The species has been suggested as a suitable bioindicator species for some heavy metals in the marine environment. Bryan *et al.* (1983) suggested that the species is a reasonable indicator for Ag, Cd, Pb and perhaps As. In the Fal estuary *Patella vulgata* occurs at, or just outside, Restronguet Point, at the end of the creek where metal concentrations are in the order: Zinc (Zn) 100-2000 µg/l, copper (Cu) 10-100µg/l and cadmium (Cd) 0.25-5µg/l (Bryan & Gibbs, 1983). However, in the laboratory *Patella vulgata* was found to be intolerant of small changes in environmental concentrations of Cd and Zn by Davies (1992). At concentrations of 10µg/l pedal mucus production and levels of activity were both reduced, indicating a physiological response to metal concentrations. Exposure to Cu at a concentration of 100 µg/l for one week resulted in progressive brachycardia (slowing of the heart beat) and the death of limpets. Zn at a concentration of 5500 µg/l produced the same effect (Marchan *et al.*, 1999).

The effects of tributyl tin (TBT), used in anti-fouling paints, on *Nucella lapillus* have been extensively documented and represent one of the best known examples of the effects of chemical pollution.

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.

Contamination by hydrocarbons, at levels greater than the pressure benchmark may adversely impact the biotope.

Himanthalia elongata survived the 'Torrey Canyon' oil spill and indeed extended 2 m vertically up the shore, due to the absence of grazers (Southward & Southward, 1978). The species lives in the lower eulittoral and sublittoral fringe, which means that oil will rapidly be washed off the fronds. It also usually occurs in areas with strong currents, allowing oil to be rapidly dispersed.

O'Brien & Dixon (1976) stated that red algae were the most sensitive group of algae to oil contamination especially in combination with dispersant contamination, possibly due to the susceptibility of the photosynthetic pigment phycoerythrin to chemical damage. Filamentous forms are considered to be most sensitive. Observations following oil spills indicate that grazing species are particularly intolerance of oil pollution. Following the *Torrey Canyon* oil spill in 1967, oil and detergent dispersants affected high shore specimens of *Corallina officinalis* more than low shore specimens. Plants in deep pools were afforded some initial protection, although probably later affected by contaminated runoff. In areas of heavy spraying, however, *Corallina officinalis* was killed (Smith 1968).

Thick layers of deposited oil would probably interfere with respiration and spoil food supplies for *Patella vulgata*. Limpets are unable to remain 'closed-off' from the environment for very long, and the adductor muscles relax occasionally, lifting the shell very slightly exposing the animal to contaminants. After the *Braer* oil spill, in common with many other oil spills, the major impact in the intertidal zone was on the population of limpets and other grazers. In West Angle Bay, where fresh oil from the *Sea Empress* tanker reached rocky shores within one day of the spill, limpet mortality was 90 % (Glegg *et al*., 1999). In the case of the *Torrey Canyon* spill the quantity and toxicity of the oil dispersants applied to the shore caused more mortalities than the oil alone, *Patella vulgata* being particularly susceptible, although all animals and many algae were killed in areas heavily sprayed (Raffaelli & Hawkins, 1996). Following oil pollution rocky shore communities are highly disturbed owing to the loss of structuring species. The recovery period can be extensive owing to both loss of species and the subsequent extreme fluctuations in abundance. In the *Torrey Canyon* incident, following the death of grazing species, a dense green flush of ephemeral algae (*Blidingia* & *Ulva*) developed and lasted for nearly a year, whilst after six months *Fucus vesiculosus* and *Fucus serratus* began to colonize the shore and persisted in dense stands for between 1 to 3 years. *Patella vulgata* colonized affected shores within the year and thrived in damp conditions under the fucoids, its grazing inhibited further extensive fucoid recruitment. Abnormal numbers of limpets accrued and cleared rocky substrata of much of the algae, allowing, after a period of 4 years (in dispersant treated areas), increased barnacle recruitment. Fucoid cover remained abnormal for the first 11 years following the spill and fluctuated for 15 years, whilst the population structure of *Patella vulgata* remained abnormal for at least 10 years (Smith, 1968; Southward & Southward, 1978; Hawkins & Southward, 1992). In longer term studies into the environmental effects of oil refinery effluent discharged into Littlewick Bay, Milford Haven, the number of limpets, usually found in substantial numbers on this type of shore, were considerably reduced in abundance on areas close to the discharge (Petpiroon & Dicks, 1982). In particular only large individuals were found close to the outfall point and juveniles were completely absent, suggesting that observed changes in abundance resulted from effluent effects on larval stages rather than upon adults directly.

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.

Contamination at levels greater than the pressure benchmark may adversely impact the biotope. No evidence was found specifically relating to the intolerance of *Himanthalia elongata* to synthetic chemicals. 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. They also report that red algae are effective indicators of detergent damage since they undergo colour changes when exposed to relatively low concentration of detergent. Smith (1968) reported that 10 ppm of the detergent BP 1002 killed the majority of specimens in 24hrs in toxicity tests. Laboratory studies of the effects of oil and dispersants on several red algal species concluded that they were all sensitive to oil/dispersant mixtures, with little difference between adults, sporelings, diploid or haploid life stages (Grandy, 1984, cited in Holt *et al*., 1995). Cole *et al.* (1999) suggested that herbicides, such as simazine and atrazine, were very toxic to macrophytes. The evidence suggests that in general red algae are very sensitive to synthetic chemicals.

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 to assess this pressure at the benchmark. Algae bioaccumulate radionuclides (with extent depending on the radionuclide and the algae species) but adverse effects have not been reported at low levels.

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

The effects of reduced oxygenation on algae are not well studied. Plants require oxygen for respiration, but this may be provided by production of oxygen during periods of photosynthesis. Lack of oxygen may impair both respiration and photosynthesis (see review by Vidaver, 1972). A study of the effects of anoxia on a red alga, *Delesseria sanguinea*, revealed that specimens died after 24 hours at 15°C but that some survived at 5°C (Hammer, 1972). This biotope would only be exposed to low oxygen in the water column intermittently during periods of tidal immersion. In addition, in areas of wave exposure and moderately strong current flow low oxygen levels in the water are unlikely to persist for very long as oxygen levels will be recharged by the incorporation of oxygen in the air into the water column or flushing with oxygenated waters.

No evidence was found to assess this pressure for the red algae turfs. However, the associated species are unlikely to be impacted by this pressure, at the benchmark. The associated invertebrate species show high tolerances for reduced oxygen at levels that exceed the pressure benchmark. *Littorina littorea* can easily survive 3-6 days of anoxia (Storey *et al*., 2013). Limpets can also survive for a short time in anoxic seawater; Grenon & Walker, (1981) found that in oxygen free water limpets could survive up to 36 hours, although Marshall & McQuaid (1989) found a lower tolerance for *Patella granularis*, which survived up to 11 hours in anoxic water. *Patella vulgata* and *Littorina littorea* and *Nucella lapillus* are able to respire in air, mitigating the effects of this pressure during the tidal cycle.

Sensitivity assessment. No direct evidence for the effects of hypoxia on red algal turfs was found. As the biotope will only be exposed to this pressure when submerged and respiration will occur in air, biotope resistance was assessed as 'High' and resilience as 'High' (no effect to recover from), resulting in an assessment of 'Not sensitive'.

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

No evidence was found to assess the sensitivity of the key characterizing species,*Himanthalia elongata*, to changes in nutrient enrichment. The red algal turfs in the biotope and encrusting corallines are considered unlikely to be negatively affected by this pressure at the benchmark based on available evidence. Overall. Littler & Littler (2013) suggest that encrusting corallines as a group can tolerate both low and elevated levels of nutrients. The key characterizing *Corallina officinalis* have been identified worldwide as species that occur in areas subject to increased nutrient input within the vicinity of sewage outfalls and at intermediately polluted sites (Bellgrove *et al*., 2010; Littler & Murray, 1975; May, 1985; Brown *et al*., 1990; Bellgrove *et al*., 1997, Arévalo *et al*., 2007). For example Kindig & Littler (1980) demonstrated that *Corallina officinalis* var. *chilensis* in South California showed equivalent or enhanced health indices, highest productivity and lowest moralities (amongst the species examined) when exposed to primary or secondary sewage effluent. *Corallina elongata* and the crusting coralline *Lithophyllum incrustans* were present at sites dominated by *Ulva* spp. in the Mediterranean exposed to high levels of nutrient enrichment from domestic sewage (Arévalo *et al*., 2007). Grazers in the biotope may benefit from increased availability of food resources, due to enhanced growth.

Atalah & Crowe (2010) added nutrients to rockpools in controlled experiments. The rockpools were occupied by a range of algae including encrusting corallines, turfs of *Mastocarpus stellatus*, *Chondrus crispus* and *Corallina officinalis* and green and red filamentous algae. The invertebrates present were mostly *Patella ulyssiponensis*, the winkle *Littorina littorea* and the flat top shell *Gibbula umbilicalis*. Nitrogen and phosphorous enhancement was via the addition of fertilisers, as either 40 g/litre or 20 g/litre. The treatments were applied for seven month and experimental conditions were maintained every two weeks. A grazer and nutrient addition treatments was also applied where grazers were removed manually from pools and a 1 m strip bordering the pools. The experimental treatments do not directly relate to the pressure benchmark but indicate some general trends in sensitivity. In the pools the chronic addition of nutrients had no significant effect on the cover of crustose coralline algae or the red turfing algae. There was a significant interactive effect of grazing and nutrients however on the cover of crustose coralline algae. Pair-wise comparisons showed that cover of crustose coralline algae was significantly reduced where nutrients were added only under reduced grazing conditions, with an absolute decrease of 44% (±8.0 S.E.) relative to the controls (P<,0.05). Grazers limit the effects of nutrient enrichment and in their absence significant changes in the structure of the algal assemblage may emerge following eutrophication.

Sensitivity assessment. The pressure benchmark is relatively protective and may represent a reduced level of nutrient enrichment in previously polluted areas. Due to the tolerance of high levels of nutrient input demonstrated generally by turf-forming species e.g. Belgrove *et al*., (2010) and Atalah & Crowe, (2010), resistance to this pressure is assessed as 'High' and resilience as 'High' so that the biotope is assessed as 'Not sensitive'. No evidence was found for the key characterizing sepcies *Himanthalia elongata* and confidence in the assessment is therefore low.

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

Where the biotope occurs in tide swept or wave exposed areas (Connor *et al.*, 2004) water movements will disperse organic matter reducing the level of exposure. As *Himanthalia elongata* are sensitive to even low levels of siltation sedimentation on bare rock may reduce recruitment success. However, Himanthalia elongata also settles within red algal turfs (Stengel *et al.*, 1999) (which may contain sediment). The associated species *Corallina officinalis* has been noted to increase in abundance and may form extensive turfs within the vicinity of sewage outfalls and at intermediately polluted sites (Belgrove *et al.*, 2010; Littler & Murray, 1975; May, 1985; Brown *et al.*, 1990). *Corallina elongata* and the crusting coralline *Lithophyllum incrustans* were present at sites dominated by *Ulva* spp. in the Mediterranean exposed to high levels of organic pollution from domestic sewage (Arévalo *et al.*, 2007). As turf forming algae including the red algal turf and particularly *Corallina* spp. trap large amounts of sediment the turf is not considered sensitive to sedimentation. The turfs host a variety of associated species and deposit feeders amongst these would be able to consume inputs of organic matter.

Sensitivity assessment. Based on resistance to sedimentation, exposure to wave action, the presence of detrital consumers and the dominance of turfs in areas subject to sewage inputs, resistance is assessed as 'High' and resilience as 'High' (by default). The biotope is therefore considered to be 'Not sensitive' to this pressure at the benchmark. As no evidence was found for the key characterizing species, *Himanthalia elongata*, confidence in the assessment is 'Low'.

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

A change to a sedimentary substratum would significantly alter the character of the biotope and would lead to the development of a biological assemblage more typical of the changed conditions. A change to an artificial substratum could also impact the development of this biotope as species may have settlement preferences for particular surface textures. An increase in the availability of hard substratum may be beneficial in areas where sedimentary habitats were previously unsuitable for colonisation e.g. coarse, mobile sediments. However, artificial hard substratum may also differ in other characteristics from natural hard substratum, so that replacement of natural surfaces with artificial may lead to changes in the biotope through changes in species composition, richness and diversity (Green *et al*., 2012; Firth *et al*., 2014) or the presence of non-native species (Bulleri & Airoldi, 2005).

Moss *et al*., 1973, surveyed shores in the north east of England and recorded the substrata types where *Himanthalia elongata* occurred. Populations occurred on a variety of substrata including quartz dolorite, coal measure sandstone, limestone and millstone grit. Horizontal ledges of mangesian limestone were not colonized as it was too friable and easily eroded to allow firm attachment. *Himanthalia elongata* was absent from outcrops that were frequently silted where silts were mobile but it could colonise mats of sediment trapping red algae particularly *Rhodochorton* spp. (Moss *et al*., 1973).

Corallina officinalis shows optimal settlement on finely rough artificial substrata (0.5 - 1mm surface particle diameter). Although spores will settle and develop as crustose bases on smooth surfaces, fronds were only initiated on rough surfaces. *Corallina officinalis* settled on artificial substrata within one week in the field in summer months in New England (Harlin & Lindbergh 1977). However, in the laboratory fronds can grow from bases attached to smooth surfaces (Wiedeman pers comm. Previous MarLIN review) Similarly, tests with stone panels fixed to the sublittoral, mid-tide and high-tide levels of varying roughness found that *Ulva* species settle preferentially on smother, fine grained substratum (chalk, mottled sandstone) and *Porphyra purpurea* on rougher, granulated substratum (limestone, granite, basaltic larvae) (Luther, 1976).

Changes in substratum type can also lead to indirect effects. For example, Shanks & Wright (1986) observed that limpet mortalities were much higher at sites where the supply of loose cobbles and pebbles were greater, leading to increased abrasion through wave action 'throwing' rocks onto surfaces. Littorinids are found on a variety of shores, including sedimentary so a change in type may not significantly affect this species and some of the invertebrate species such as nematodes, amphipods and oligochaetes and polychaetes associated with sediments trapped in the algal turf are also found in sedimentary habitats

Sensitivity assessment. A change to a soft sedimentary habitat would remove the habitat for this biotope, resistance is assessed as ‘None’ and resilience as ‘Very Low’ as the change is considered to be permanent. Sensitivity is therefore 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

Not relevant to biotopes occurring on bedrock.

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

The species characterizing this biotope are epiflora and epifauna 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	Medium Q: High A: High C: Low	High Q: High A: Low C: Medium	Low Q: High A: Low C: Low

The species characterizing this biotope occur on the rock and therefore have no shelter from abrasion at the surface. No direct evidence was found to assess abrasion impacts on the key characterizing species *Himanthalia elongata*. It is possible that this species is vulnerable to abrasion as the attachment area to the surface is small in comparison with red algal turfs arising from crustose bases that cover a larger surface area and are flat and tough.

Abrasion can remove the frond of red and brown algae although experimental results vary. Brosnan & Crumrine (1994) noted that trampling significantly reduced algal cover within 1 month of trampling. Foliose algae were particularly affected and decreased in cover from 75% to 9.1% in trampled plots. *Mastocarpus papillatus* decreased in abundance from 9% to 1% in trampled plots but increased in control plots. *Fucus distichus* decreased in the summer months only to recover in winter but in trampled plots remained in low abundance (between 1 and 3% cover). Trampling resulted in a decrease in cover of *Pelvetiopsis limitata* from 16% to 1.5%. However, after trampling ceased, recovery of algal cover including *Iridaea cornucopiae* and *Mastocarpus papillatus* was rapid (ca 12 months) (Brosnan & Crumrine, 1994). Schiel & Taylor (1999) also observed a decrease in understorey algae (erect and encrusting corallines) after 25 or more tramples, probably due to an indirect effect of increased desiccation as above. However, Schiel & Taylor (1999) did not detect any variation in other algal species due to trampling effects. Similarly, Keough & Quinn (1998) did not detect any effect of trampling on algal turf species.

Littler & Littler, 1984) suggest that the basal crustose stage of *Corallina officinalis* is adaptive to resist sand scour and wave shearing (as well as physiological stressors such as desiccation and heating). The base is much tougher than the fronds, as shown by experiments that demonstrate that the base has nearly twice the mechanical resistance (measured by penetration) of fronds (Littler & Kauker, 1984). In general, studies show that *Corallina* and other turf forming algae appear to be relatively resistant to single events and low levels of trampling. Brosnan & Crumrine (1994), for example, found that in experimentally trampled plots the cover of foliose and canopy forming species declined while turf forming algae were relatively resistant. Similarly, a comparison of rocky intertidal ledges that received different amounts of visitors in Dorset, England, found that *Corallina officinalis* were present on both heavily visited and less visited ledges suggesting that this species has some resistance to trampling (Pinn & Rodgers, 2005). Povey & Keough (1991) in Mornington Peninsula, Australia investigated the effects of sustained trampling on intertidal coralline algal mats where upright branching *Corallina* spp. formed a turf with other red algae with sand and encrusting coralline algae between turfs. The experimental strips were 2 m long and 0.5 m wide. The percentage cover of upright *Corallina* spp. was significantly affected by 25 passages of a strip per day after 12 and 33 days. The algae appeared flattened and were shorter (1-2 cm high) compared with the low intensity and control plots (3-4 cm high). However, low intensity trampling within a strip (2 passages/ day) did not significantly affect the coralline turf. Brown & Taylor (1999) found that higher intensities of trampling damaged turfs. Moderate (50 steps per 0.09 m^2) or more trampling on intertidal articulated coralline algal turf in New Zealand reduced turf height by up to 50 % and reduced the weight of sand trapped within turf to about one third of controls. This resulted in declines in densities of the meiofaunal community within two days of trampling. Although the community returned to normal levels within 3 months of trampling events, it was suggested that the turf would take longer to recover its previous cover (Brown & Taylor, 1999). Similarly, Schiel & Taylor (1999) noted that trampling had a direct detrimental effect on coralline turf species on the New Zealand rocky shore. At one site coralline bases were seen to peel from the rocks (Schiel & Taylor, 1999), however, this was probably due to increased desiccation caused by loss of the algal canopy.

The barnacles, limpets and littorinids that occur in this biotope, have some protection from hard shells or plates but abrasion may damage and kill individuals or detach these. All removed barnacles would be expected to die as there is no mechanism for these to reattach. Removal of limpets and barnacles may result in these being displaced to a less favourable habitat and injuries to foot muscles in limpets may prevent reattachment. Although limpets and littorinids may be able to repair shell damage, broken shells while healing will expose the individual to more risk of desiccation and predation. Evidence for the effects of abrasion are provided by a number of experimental studies on trampling (a source of abrasion) and on abrasion by wave thrown rocks and pebbles.

The effects of trampling on barnacles appears to be variable with some studies not detecting significant differences between trampled and controlled areas (Tyler-Walters & Arnold, 2008). However, this variability may be related to differences in trampling intensities and abundance of populations studied. The worst case incidence was reported by Brosnan & Crumrine (1994) who found 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 (*Semibalanus glandula* and *Chthamalus dalli*) 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/hr) led to reductions in barnacle cover. Single step experiments provide a clearer, quantitative indication of sensitivity to single events of 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, (*Chthamlus antennatus* about 3mm long), were crushed. However, on the same shore, the authors found that limpets may be relatively more resistant to abrasion from trampling. Following step and kicking experiments, few individuals of the limpet C*ellana trasomerica*, (similar size to *Patella vulgata*) suffered damage or relocated (Povey & Keough, 1991). One kicked limpet (out of 80) was broken and 2 (out of 80) limpets that were stepped on could not be relocated the following day (Povey & Keough, 1991). On the same shore less than 5% of littorinids were crushed in single step experiments (Povey & Keough, 1991).

Shanks & Wright (1986), found that even small pebbles (<6 cm) that were thrown by wave action in Southern California shores could create patches in aggregations of the barnacle, *Chthamalus fissus*, and could smash owl limpets (*Lottia gigantea*). Average, estimated survivorship of limpets at a wave exposed site, with many loose cobbles and pebbles allowing greater levels of abrasion was 40% lower than at a sheltered site. Severe storms were observed to lead to almost total destruction of local populations of limpets through abrasion by large rocks and boulders. In sites with mobile cobbles and boulders increased scour results in lower densities of *Littorina* spp. compared with other, local sites with stable substratum (Carlson *et al*., 2006).

Sensitivity assessment. The impact of surface abrasion will depend on the footprint, duration and magnitude of the pressure. Resistance, to a single abrasion event of *Himanthalia elongata*, red algal turf and associated barnacles, limpets and littorinids is assessed as ‘Medium’ (<25% of population damaged or removed) as some damage to individuals and fronds of algal turfs may be removed although the bases may remain. Recovery is ‘High’, through regrowth of fronds so that sensitivity is assessed as ‘Low’. Resistance and resilience will be lower (and hence sensitivity greater) to abrasion events that exert a greater crushing force and remove the bases of algal turfs and damage and remove a greater proportion of species.

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 epiflora and epifauna occurring on rock which is resistant to subsurface penetration. The assessment for abrasion at the surface only is therefore considered to equally represent sensitivity to this pressure.

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

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). The biotope occurs in shallow waters at the littoral/sublittoral fringe where light attenuation due to increases in turbidity is probably low. Red algae and coralline algae especially are known to be shade tolerant and are common components of the understorey on seaweed dominated shores. Therefore, a decrease in light intensity is unlikely to adversely affect the biotope.

Scour from suspended particles induces high mortality in early post settlement (EPS) algal stages and prevents the settlement of propagules owing to accumulation of silt on the substratum (Vadas *et al*., 1992). Young stages of *Himanthalia elongata* were found to be very intolerance of resultant siltation (Moss *et al*., 1973), hence the impact of increased suspended sediment would depend on the time of year. If increased siltation occurred from June to December, when gametes are released, the population would be highly intolerant because zygotes cannot grow on silt (Moss *et al*., 1973). The fronds of more delicate species such as *Palmaria palmata* may also be damaged by scour. On sites affected by high levels of resuspended colliery waste particles, Hyslop *et al*., (1997) found that *Palmaria palmata* and *Ulva* spp. were reduced or absent, although the more tough fucoids were less affected. It is not clear how the levels of suspended solids experienced by these sites relate to the pressure benchmark. Coralline algae, especially the crustose forms are thought to be resistant of sediment scour (Littler & Kauker, 1984), and will probably not be adversely affected at the benchmark level.

Increases in the cover of sediment trapping, turf forming algae at the expense of canopy forming species has been observed worldwide in temperate systems and has been linked to increased suspended solids linked to human activities worldwide (Airoldi, 2003). *Corallina* species accumulate more sediment than any other alga (Hicks, 1985). Hence an increase in suspended sediment is likely to accumulate in the coralline turf. In the Bay of Fundy for example, 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.

An accumulation of sediment within the turf may attract more sediment dwelling interstitial invertebrates such as nematodes, harpacticoids and polychaetes, although in more wave exposed locations accumulation of sediment is likely to be minimal.

Increased suspended sediment may reduce growth rate in *Semibalanus balanoides* due to the energetic costs of cleaning sediment particles from feeding apparatus although if the organic content is high suspension feeders could also benefit. *Patella vulgata* and *Mytilus edulis* are considered to tolerate an increase in suspended sediment as they are found in turbid estuaries where suspended sediment levels are high. It should be noted however that *Patella vulgata* is very sensitive to even low levels of siltation (see siltation pressure).

This community is unlikely to be dependent on suspended sediment. Although accumulated sediment within coralline turf habitats is likely to increase the species diversity of the epiphytic fauna, in very wave exposed locations, accumulated sediment in the habitat is likely to be minimal. A reduction in suspended sediment will probably reduce the risk of scour, and reduce food availability for the few suspension feeding species in the biotope (e.g. barnacles and spirorbids present), although effects are not likely to be lethal. An increase in light intensity is unlikely to adversely affect the biotope as plants can acclimate to different light levels.

Sensitivity assessment. The biotope is considered to be ‘Not sensitive’ to a reduction in suspended solids, although this may reduce food supply to the barnacles and other filter feeders that occur in this biotope. An increase in suspended solids may lead to some sub-lethal abrasion of fronds however, evidence globally indicates that increases in suspended solids favour the turf-forming algae that characterize this biotope (Airoldi, 2003). The encrusting corallines also seem to be tolerant. No evidence was found to assess sensitivity of *Himanthalia elongata*, however, buttons can slough off sediments and the thallus is tough so unlikely to be abraded or covered by settling sediments. Resistance is therefore assessed as ‘High’ and resilience as ‘High’ (by default) so that the biotope is considered to be ‘Not sensitive’. Over a period longer than a year, increased turbidity and scour may result in some changes to the species composition within the biotope due to reduced recruitment and turbidity, with replacement of thinner, foliose red algae with more scour tolerant algae such as *Ahnfeltia plicata* and *Rhodothamniella floridula*. This may also reduce recruitment by *Himanthalia elongata* and lead to biotope reclassification.

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

Sedimentation can directly affect assemblages inhabiting rocky shores in different ways, particularly by the burial/smothering and scour/abrasion of organisms. This biotope occurs in habitats that are moderately exposed or exposed to wave action (Connor *et al*., 2004). In areas with greater water flow or wave action, excess sediments will be removed from the rock surface within a few tidal cycles, reducing the time of exposure to this pressure.

The state of the tide will mediate the degree of impact on macroalgae. If smothering occurs at low tide when the algae is lying flat on the substratum, then most of the organism as well as the associated community will be covered by the deposition of fine material at the level of the benchmark. Smothering will prevent photosynthesis resulting in reduced growth and eventually death. If however smothering occurs whilst the alga is submerged standing upright then the photosynthetic surfaces of adult plants will be left uncovered. The resistance of this biotope to the given pressure may vary with time of day. Germlings however are likely to be smothered and killed in both scenarios and are inherently most susceptible to this pressure. Indeed early life stages are smaller in size than adults and are thus most vulnerable to this pressure as even a small load of added sediment will lead to the complete burial. In general, propagules, early post-settlement stages and juveniles suffer severe stress and mortality from sediments (Devinny & Volse, 1978; Eriksson & Johansson, 2003; Berger *et al*. 2003; Vadas *et al*., 1992; Airoldi, 2003). Moss *et al*., (1973), found that growth of zygotes of *Himanthalia elongata* were inhibited by a layer of silt 1-2 mm thick and that attachment on silt was insecure. However, Stengel *et al*., (1999) found *Himanthalia elongata* at Finavarra (West Ireland) growing on flat limestone under shallow drifting sand. Intermittent exposure to silt and coarser sediments may therefore be tolerated by adult plants.

Increased abundance of algal turfs worldwide has been linked to sediment perturbations although not all the pathways and mechanisms of these effects are clear (see review by Airoldi, 2003). However, even the most tolerant of organisms would eventually suffer from inhibition and mortality following smothering although the thresholds for these effects have has not been identified (Airoldi, 2003). *Corallina officinalis* and others within the genus (e.g. *Corallina pinnatifolia* and *Corallina vancouveriensis*) are found on shores subject to high rates of sedimentation that are periodically disturbed by sand burial and scour (Stewart, 1989). Coralline turfs also trap sediments within the turf. The amount of sediment present and the associated fauna varies naturally depending on local conditions such as wave exposure (Dommasnes, 1969). On intertidal shores in southern California the amount of sediment trapped within turfs of *Corallina* spp. varied seasonally from < 5mm to >4.5 cm and was closely related to species composition and the structure of the turf. Airoldi (2003) identified a number of morphological, physiological and life history traits that conferred high levels of tolerance to sedimentation. Those shared by *Corallina* spp are the regeneration of upright fronds from a perennial basal crust resistant to burial and scour, calcified thalli, apical meristems, large reproductive ouptuts, lateral vegetative growth and slow growth rates (Airoldi, 2003). Experimental deposition of sand on coralline turfs and maintained at 3 cm or 6 cm for one month via daily top-ups did not remove the turfs but did lead to rapid (within 1 hours changes in the invertebrate species as highly mobile species moved away from the turf with later colonisation by sand adapted species (Huff & Jarett, 2007). The community had recovered one month after sand deposition ceased (Huff & Jarett, 2007).

The associated species, *Patella vulgata* and *Littorina* spp. are likely to be negatively affected by siltation (Airoldi & Hawkins, 2007; Chandrasekara & Frid,1998; Albrecht & Reise, 1994). Experiments have shown that the addition of even thin layers of sediment (approximately 4 mm) inhibit limpet grazing and result in loss of attachment and death after a few days Airoldi & Hawkins (2007). The laboratory experiments are supported by observations on exposed and sheltered shores with patches of sediment around Plymouth in the south west of England as *Patella vulgata* abundances were higher where deposits were absent (Airoldi & Hawkins, 2007). Littler *et al*., (1983) found that the another limpet species, *Lottia gigantea* on southern Californian shores was restricted to refuges from sand burial on shores subject to periodic inundation by sands.

Sensitivity assessment. Deposition of 5 cm of fine material (see benchmark) in a single incident is unlikely to result in significant mortality before sediments are removed by current and wave action. Burial may lower survival and germination rates of spores and cause some mortality in early life stages of *Fucus serratus* and foliose red algae. Adults are more resistant but will experience a decrease in growth and photosynthetic rates. Eg. depending on how long the sediment remains. Overall resistance is assessed as 'High and resilience as ‘High’ (by default).

Overall the biotope is considered to be 'Not sensitive' to smothering at the level of the benchmark on the basis that smothering sediment is rapidly removed due to wave action or tidal streams. Where wave action and currents are reduced mortality may be greater and resistance would be lower. It should be noted that the associated *Patella vulgata* and littorinids have higher sensitivities to this pressure and that removal of these species may result in changes in algal composition and abundances. Increased chronic siltation may lead to changes in assemblage and loss of *Himanthalia elongata* leading to biotope reclassification.

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

Sensitivity to this pressure will be mediated by site-specific hydrodynamic conditions and the footprint of the impact. Where a large area is covered sediments may be shifted by wave and tides rather than removed. However, mortality will depend on the duration of smothering, where wave action rapidly mobilises and removes fine sediments, survival may be much greater. No evidence was found to assess this pressure at the benchmark. A deposit at the pressure benchmark would cover species with a thick layer of fine materials, however, as this biotope occurs in the lower intertidal it is subject to prolonged immersion and silts may be relatively rapidly re-suspended and removed.

Species associated with this biotope such as limpets and littorinids would not be able to escape from deposits and would likely suffer mortality (see evidence for light siltation). Even the most tolerant of organisms would eventually suffer from inhibition and mortality following smothering although the thresholds for these effects have has not been identified (Airoldi, 2003).

Sensitivity assessment. At the level of the benchmark (30 cm of fine material added to the seabed in a single event) smothering is likely to result in mortalities of macroalgae, particularly germlings, and invertebrate grazers. Resistance is assessed as ‘Low’ as many individuals exposed to siltation at the benchmark level are predicted to die and resilience is assessed as ‘Medium’. Overall the biotope has a ‘Medium’ sensitivity to siltation at the pressure benchmark, based on rapid recovery.

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

No evidence.

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

Not relevant.

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

Moss *et al*. (1973) tested the effects of different levels of light intensity on the growth of early stages of *Himanthalia elongata*. Zygotes were plated and grown under light levels varying from darkness to 6000 Lux at three temperatures; 5, 10 and 20 °C. Germination was measured after 7 days and growth after 1 month. In total darkness a small proportion of zygotes germinated at 10 and 20 °C (5-17 % and 0-6 % respectively). At 5°C and 20°C germination rates were similar across all light intensities, highest germination rates were achieved at the highest light intensity at 10 °C (40-91 %). Maximum growth occurred at 10 °C and light intensities between 1272 and 5936 Lux (Moss *et al*., 1973)

Corallina officinalis and other red algae are shade tolerant, often occurring under a macralgal canopy that reduces light penetration. Other red algae in the biotope are flexible with regard to light levels. Canopy removal experiments in a rocky sub tidal habitat in Nova Scotia, Canada by Schmidt & Scheibling (2007) did not find a shift in understorey macraoalgal turfs (dominated by *Corallina officinalis*, *Chondrus crispus* and *Mastocarpus stellatus*) to more light-adapted species over 18 months.

Sensitivity assessment. As the key structuring and characterizing species *Himanthalia elongata* shows maximal growth at a range of light temperatures and the associated red algae can colonize a broad range of light environments from intertidal to deeper sub tidal and shaded understorey habitats the biotope is considered to have 'High' resistance and, by default, 'High' resilience and is therefore judged to be 'Not sensitive' to this pressure.

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

The key characterizing species *Himanthalia elongata* has limited dispersal with populations recruiting from surrounding adults. As this species does not depend on external sources of propagules to sustain the population this pressure is assessed as 'Not relevant'

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

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

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

Not relevant.

Biological Pressures

	Resistance	Resilience	Sensitivity

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

Key characterizing species within this biotope are not cultivated or translocated. This pressure is therefore considered 'Not relevant' to this biotope group. No information was found on current production of *Mastocarpus stellatus*, *Chondrus crispus* or other turf forming red seaweeds in the UK and it is understood that wild harvesting rather than cultivation is the method of production for these and littorinids. No evidence was found for the effects of gene flow between cultivated species and wild populations. Although cultivation of different genotypes may lead to gene flow between wild and cultivated populations the limited dispersal may reduce exposure. Some negative effects may arise from hybridisation between very geographically separated populations but there is no evidence to suggest that gene flow between different UK haplotypes would lead to negative effects. This pressure is therefore considered 'Not relevant' to this biotope group.

The key characterizing species *Palmaria palmata* may be cultivated for use as biofilters and/or food. No information was found on current production in the UK and no evidence was found for the effects of gene flow between cultivated species and wild populations. Experiments by van der Meer (1987) found that the hybrids of *Palmaria palmata* crosses from Canada and Ireland had vigorous growth and normal morphology, however the tetraspores had lower viability and those that germinated produced abnormal sporelings. It was concluded that populations from Ireland and Canada represent the same species that is in the process of splitting into sibling species. Populations around the UK express different haplotypes (Provan *et al*., 2005) indicating some genetic variation between populations. Although cultivation of different genotypes may lead to gene flow between wild and cultivated populations the limited dispersal may reduce exposure. Some negative effects may arise from hybridisation between very geographically separated populations but there is no evidence to suggest that gene flow between different UK haplotypes would lead to negative effects.

Sensitivity assessment. No direct evidence for genetic effects arising from cultivation and/or translocation was found, resistance is assessed as 'High' and resilience as 'High' so that the biotope is 'Not sensitive'. This assessment may require updating if cultivation methods change and translocate material from geographically separate locations.

Introduction or spread of invasive non-indigenous species	Low Q: High A: Low C: Low	Very Low Q: Low A: NR C: NR	High Q: High A: Low C: Low

Invasive non-indigenous species (INIS) that can alter habitats (ecological engineers), or out-compete native macroalgae for space and other resources such as light and nutrients, are the most likely species to negatively affect this biotope. Space pre-emption by encrusting corallines and the crustose bases of the macroalgae forming the turf, as well as the trapped sediment within the turf, may prevent settlement of INIS until disturbance events create gaps for invasion. However, in the Mediterranean crustose corallines and algal turfs facilitate attachment of *Caulerpa racemosa* by providing a more complex substratum than bare rock (Bulleri & Benedetti-Cecchi, 2008).

The first records of the INIS *Watersipora subtorquata* in German coastal waters of the North Sea arose from individuals attached to drifting or stranded *Himanthalia elongata* that probably drifted from the English Channel 200 miles away (Kuhlenkamp & Kind, 2013). These records indicate that

Himanthalia elongata can be colonized by this species. Although no evidence for negative effects has been reported.

Algal species which may have overlapping habitat requirements include the green seaweed *Codium fragile* subsp *tormentosoides* (now renamed as *Codium fragile fragile*) and the red seaweed *Heterosiphonia japonica*, neither of these have so far been recorded in nuisance densities (Sweet, 2011j). Wireweed, *Sargassum muticum*, grows best on sheltered shores and in rockpools (Sewell, 2011c) and the wave exposed habitats where this biotope occurs may not be suitable for establishment. The red seaweeds *Heterosiphonia japonica* and *Neosiphonia harveyi* may also occur in this biotope but again no impacts have been reported.

The morphologically similar, red seaweed *Grateloupia turuturu* occurs on the lower shore in pools. No ecosystem impacts have been reported in Great Britain; however this large, fast-growing seaweed may have the potential to displace native seaweed species and shade neighbouring species. In North America this species is a major competitor of *Chondrus crispus* which provides an important winter food source for littorinids and other invertebrates. As *Grateloupia turuturu* dies-back in the winter, the dispolacement of other species may therefore affect grazers (Sweet, 2011g).

The tunicates *Didemnum vexillum* and *Asterocarpa humilis*, the hydroid *Schizoporella japonica* and the bryozoan *Watersipora subatra* (Bishop, 2012c, Bishop, 2015a and b; Wood, 2015) are currently only recorded from artificial hard substratum in the UK and it is not clear what their established range and impacts in the UK would be. *Didemnum vexillum* occurs in tide pools in other areas where it has become established (Bishop, 2012c) and can have substantial effects on communities, similarly the tunicates *Corella eumyota* and *Botrylloides violaceus* can smother rock habitats (Bishop, 2011b and 2012b). A significant potential INIS is the Pacific oyster *Magallana gigas*, as its distribution and environmental tolerances are considered to overlap with this biotope and this reef forming species can alter habitat structure. This species may also affect the grazers present in the biotope. No evidence was found for effects on rock pools, although on the Mediterranean coast *Magallana gigas* is cultivated in micro-tidal lagoons and has established wild populations (Miossec *et al*., 2009, cited from Herbert *et al*., 2012). In the Wadden Sea and North Sea, *Magallana gigas* overgrows mussel beds in the intertidal zone (Diederich, 2005, 2006; Kochmann *et al*., 2008), although larvae did show preference for settling on conspecifics before the mussels and struggled to settle on mussels with a fucoid covering. It has been observed that mussel beds in the Wadden Sea that are adjacent to oyster farms were quickly converted to oyster beds (Kochmann *et al*., 2008).

The non-native crab *Hemigrapsus sanguineus* has recently been recorded in the UK (Sweet & Sewell, 2014) and has the potential to be a significant predator of intertidal invertebrates. Significant reductions in common shore crab abundance and mussel density have been reported where the Asian shore crab has achieved high densities in mainland Europe (Sweet & Sewell, 2014). In Rye, New York, declines of approximately 80% of *Littorina littorea* in the intertidal were reported to coincide with an expansion of the *Hemigrapsus sanguineus* population (Kraemer *et al*., 2007). This crab occurs on exposed shores and may therefore occur in this biotope when established.

Sensitivity assessment. Little evidence was found to assess the impact of INIS on this biotope and much of the evidence comes from intertidal habitats in other countries. The conversion of this biotope to a *Magallana gigas* reef would present a significantly negative impact. Replacement of red algal turfs by other similar species may lead to some subtle effects on local ecology but at low abundances the biotope would still be recognisable from the description. Based on *Magallana gigas* resistance to this pressure is assessed as 'Low'. The biotope will only recover if these species are removed, either through active management or natural processes. To recognise that recovery may be prolonged, resilience is assessed as 'Very Low' and sensitivity is therefore assessed as 'High'.

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

The fronds of *Himanthalia elongata* frequently bear algal epiphytes and endophytes and a number of marine fungi but more rarely the parasite *Halosacciocolax kjellmanii* Lund (Guiry). Galls are produced by nematodes, copepods and bacteria (Irvine, 1983). The detrimental effects (if any) of such organisms are not known. The fungal pathogen, *Petersenia palmaria* n. sp (Oomycetes) which infects *Palmaria mollis*, does not affect the red algae *Palmaria palmata* (Van der Meer & Pueschel, 1985) which occurs in his biotope. Other species associated with this biotope such as littorinids, patellid limpets and other algae experience low levels of infestation by pathogens but mass-mortalities have not been recorded.

Sensitivity assessment. Based on the lack of evidence for major pathogens or significant mortalities of *Palmaria palmata* this biotope is considered to have 'High' resistance and hence 'High' resilience and is classed as 'Not sensitive' at the pressure benchmark.

Removal of target species	Low Q: High A: High C: Medium	High Q: High A: High C: Medium	Low Q: High A: High 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. Incidental removal of the key characterizing species and associated species would alter the character of the biotope.

Himanthalia elongata is harvested in the UK and Ireland for use as food, food products and cosmetics and pharmaceuticals (www.seaweedindustry.com; Stengel *et al*., 1999). The red algae *Mastocarpus stellatus* and *Chondrus crispus* are both harvested commercially in Scotland and Ireland, the stipe is removed but the base is left intact to allow the algae to re-grow. *Palmaria palmata* and *Osmundea pinnatifida* are also collected by hand commercially and recreationally for consumption. As the key characterizing and structuring species extensive removal of *Mastocarpus stellatus* and *Chondrus crispus* would alter the character of the biotope. The effect of harvesting *Chondrus crispus* has been best studied in Canada. Prior to 1980, the seaweed beds of Prince Edward Island were dominated by *Chondrus crispus* and the species was heavily exploited. Recently, there has been a marked increase in abundance of another red seaweed, *Furcellaria lumbricalis*, which is not harvested, and an associated decline in abundance of *Chondrus crispus* (Sharp *et al*., 1993). The authors suggested that harvesting has brought about the shift in community structure. Sharp *et al*. (1986) reported that the first drag rake harvest of the season, on a Nova Scotian *Chondrus crispus* bed, removed 11% of the fronds and 40% of the biomass. Efficiency declined as the harvesting season progressed. Chopin *et al*. (1988) noted that non-drag raked beds of *Chondrus crispus* in the Gulf of St Lawrence showed greater year round carposporangial reproductive capacity than a drag raked bed. In the short-term, therefore, harvesting of *Chondrus crispus* may remove biomass and impair reproductive capacity, while in the long-term, it has the potential to alter community structure and change the dominant species. Removal of other associated algae such as *Palmaria palmata* and *Osmundea pinnatifida* will reduce cover of turf forming red algae in this biotope. *Palmaria palmata* (known as dulse) is harvested from the wild for human consumption both commercially and recreationall. Garbary *et al*., (2012) studied harvested and non-harvested shores in Nova Scotia, Canada containing stands of *Palmaria palmata*. They also conducted 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).

The winkle *Littorina littorea* and the limpet *Patella vulgata* occur in low densities in this biotope and may be gathered by hand. Changes in grazer abundance can alter the character of the assemblage. Grazer removal (manual removal of all gastropods in pool and a 1m surrounding perimeter) caused strong and highly significant changes in assemblage structure in rockpools that contained red turf forming algae mainly due to an increase in the cover of green filamentous algae and a decrease in cover of live crustose coralline algae (25.40%) (Atalah & Crowe, 2010). Similarly, *Himanthalia elongata*, *Fucus serratus* and other brown seaweeds colonized new areas of Lough Hyne (Northern Ireland) following the mass mortality of the grazer *Paracentrotus lividus* (Trowbridge *et al*., 2013), demonstrating the importance of grazers to this biotope.

The removal of kelps or fucoids from this biotope would reduce shading and is not considered to negatively affect this biotope (for assessment of removal of the key characterizing species as by-catch, see the removal of non-target species pressure).

Sensitivity assessment. The species that are harvested in this biotope are all attached, sedentary or slow moving and relatively conspicuous. A single event of targeted harvesting could therefore efficiently remove individuals and resistance is assessed as ‘Low’. This assessment is supported by evidence from Sharp *et al*., (1993) on the proportion of biomass of *Mastocarpus stellatus* and *Chondrus crispus* removed commercially. Resilience of the turf forming red seaweeds is assessed as ‘High’ (based on evidence for recovery from harvesting that did not damage the algal bases although see caveats in the resilience section) and biotope sensitivity is assessed as ‘Low'. This assessment refers to a single collection event, long-term harvesting over wide spatial scales will lead to greater impacts, with lower resistance and longer recovery times. Intense harvesting of littorinids, coupled with removal of limpets would be likely to result in enhance growth of ephemeral algae based on Atalah & Crowe (2010).

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

Incidental removal of the key characterizing species and associated species would alter the character of the biotope. The biotope is characterized by *Himanthalia elongata* and red seaweeds with associated algae and invertebrates. The loss of the biological assemblage due to incidental removal as by-catch would therefore alter the character of the habitat and result in the loss of species richness. The ecological services such as primary and secondary production and the habitat provided by these species would also be lost.

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

Bibliography

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

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

Albrecht, A. & Reise, K., 1994. Effects of *Fucus vesiculosus* covering intertidal mussel beds in the Wadden Sea. *Helgoländer Meeresuntersuchungen*, **48** (2-3), 243-256.

Almada-Villela P.C., 1984. The effects of reduced salinity on the shell growth of small *Mytilus edulis* L. *Journal of the Marine Biological Association of the United Kingdom*, **64**, 171-182.

Anala, J., 1974. *Foraging strategies of two marine invertebrates.* , Ph.D. thesis, University of New Hampshire, Durham, USA.

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.

Atalah, J. & Crowe, T.P., 2010. Combined effects of nutrient enrichment, sedimentation and grazer loss on rock pool assemblages. *Journal of Experimental Marine Biology and Ecology*, **388** (1), 51-57.

Bailey, J., Parsons, J. & Couturier, C., 1996. Salinity tolerance in the blue mussel, *Mytilus edulis*. *Rep. Report no. 0840-5417, Aquaculture Association of Canada*, New Brunswick, Canada

Bamber, R.N. & Irving, P.W., 1993. The *Corallina* run-offs of Bridgewater Bay. *Porcupine Newsletter*, **5**, 190-197.

Barnes, M., 1989. Egg production in Cirripedia. *Oceanography and Marine Biology: an Annual Review*, **27**, 91-166.

Bayne, B.L. & Scullard, C., 1978. Rates of feeding by *Thais* (*Nucella*) *lapillus* (L.). *Journal of Experimental Marine Biology and Ecology*, **32**, 75-94.

Bellgrove, A., Clayton, M.N. & Quinn, G., 1997. Effects of secondarily treated sewage effluent on intertidal macroalgal recruitment processes. *Marine and Freshwater Research*, **48** (2), 137-146.

Bellgrove, A., McKenzie, P.F., McKenzie, J.L. & Sfiligoj, B.J., 2010. Restoration of the habitat-forming fucoid alga *Hormosira banksii* at effluent-affected sites: competitive exclusion by coralline turfs. *Marine Ecology Progress Series*, **419**, 47-56.

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

Berger, R., Henriksson, E., Kautsky, L. & Malm, T., 2003. Effects of filamentous algae and deposited matter on the survival of *Fucus vesiculosus* L. germlings in the Baltic Sea. *Aquatic Ecology*, **37** (1), 1-11.

Bhattacharya, D., 1985. The demography of fronds of *Chondrus crispus* Stackhouse. *Journal of Experimental Marine Biology and Ecology*, **91**, 217-231.

Bird, N.L., Chen, L.C.-M. & McLachlan, J., 1979. Effects of temperature, light and salinity of growth in culture of *Chondrus crispus*, *Furcellaria lumbricalis*, *Gracilaria tikvahiae* (Gigartinales, Rhodophyta), and *Fucus serratus* (Fucales, Phaeophyta). *Botanica Marina*, **22**, 521-527.

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

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

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

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

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

Blinks, L.R., 1955. Accessory pigments and photosynthesis. In *Photophysiology, Vol. 1.*,(ed. A.C. Giese), pp. 199-221. New York & London: Academic Press.

Boaventura, D., Alexander, M., Della Santina, P., Smith, N.D., Re, P., da Fonseca, L.C. & Hawkins, S.J., 2002. The effects of grazing on the distribution and composition of low-shore algal communities on the central coast of Portugal and on the southern coast of Britain. *Journal of Experimental Marine Biology and Ecology*, **267** (2), 185-206.

Boller, M.L. & Carrington, E., 2006. In situ measurements of hydrodynamic forces imposed on *Chondrus crispus* Stackhouse. *Journal of Experimental Marine Biology and Ecology*, **337** (2), 159-170.

Boller, M.L. & Carrington, E., 2007. Interspecific comparison of hydrodynamic performance and structural properties among intertidal macroalgae. *Journal of Experimental Biology*, **210** (11), 1874-1884.

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

Boney, S.H., 1978. Survival and growth of alpha-spores of *Porphyra schizophylla* Hollenberg (Rhodophyta: Bangiophyceae). *Journal of Experimental Marine Biology and Ecology*, **35**, 7-29.

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

Brenchley, J., Raven, J. & Johnston, A., 1996. A comparison of reproductive allocation and reproductive effort between semelparous and iteroparous fucoids (Fucales, Phaeophyta). Hydrobiologia, 326-327 (1), 185-190.

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.

Brown, P.J. & Taylor, R.B., 1999. Effects of trampling by humans on animals inhabiting coralline algal turf in the rocky intertidal. *Journal of Experimental Marine Biology and Ecology*, **235**, 45-53.

Brown, V., Davies, S. & Synnot, R., 1990. Long-term monitoring of the effects of treated sewage effluent on the intertidal macroalgal community near Cape Schanck, Victoria, Australia. *Botanica Marina*, **33** (1), 85-98.

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

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

Bryan, G.W., Langston, W.J., Hummerstone, L.G., Burt, G.R. & Ho, Y.B., 1983. An assessment of the gastropod *Littorina littorea* (L.) as an indicator of heavy metal contamination in United Kingdom estuaries. *Journal of the Marine Biological Association of the United Kingdom*, **63**, 327-345.

Bulleri, F. & Airoldi, L., 2005. Artificial marine structures facilitate the spread of a non-indigenous green alga, *Codium fragile* ssp. *tomentosoides*, in the north Adriatic Sea. *Journal of Applied Ecology*, **42** (6), 1063-1072.

Bulleri, F. & Benedetti-Cecchi, L., 2008. Facilitation of the introduced green alga *Caulerpa racemosa* by resident algal turfs: experimental evaluation of underlying mechanisms. *Marine Ecology Progress Series*, **364**, 77-86.

Burdin, K.S. & Bird, K.T., 1994. Heavy metal accumulation by carrageenan and agar producing algae. *Botanica Marina*, **37**, 467-470.

Burrows, E.M. & Lodge, S.M., 1950. Note on the inter-relationships of *Patella*, *Balanus* and *Fucus* on a semi-exposed coast. *Report of the Marine Biological Station, Port Erin*, **62**, 30-34.

Burrows, M.T. & Hughes, R.N., 1989. Natural foraging of the dogwhelk, *Nucella lapillus* (Linnaeus); the weather and whether to feed. *Journal of Molluscan Studies*, **55** (2), 285-295.

Carlson, R.L., Shulman, M.J. & Ellis, J.C., 2006. Factors Contributing to Spatial Heterogeneity in the Abundance of the Common Periwinkle *Littorina Littorea* (L.). *Journal of Molluscan Studies*, **72** (2), 149-156.

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

Chandrasekara, W.U. & Frid, C.L.J., 1998. A laboratory assessment of the survival and vertical movement of two epibenthic gastropod species, *Hydrobia ulvae*, (Pennant) and *Littorina littorea* (Linnaeus), after burial in sediment. *Journal of Experimental Marine Biology and Ecology*, **221**, 191-207.

Chopin, T., Pringle, J.D. & Semple, R.E., 1988. Reproductive capacity of dragraked and non-dragraked Irish moss (*Chondrus crispus* Stackhouse) beds in the southern Gulf of St Lawrence. *Canadian Journal of Fisheries and Aquatic Sciences*, **45**, 758-766.

Coelho, S.M., Rijstenbil, J.W. & Brown, M.T., 2000. Impacts of anthropogenic stresses on the early development stages of seaweeds. *Journal of Aquatic Ecosystem Stress and Recovery*, **7** (4), 317-333.

Cole, S., Codling, I.D., Parr, W. & Zabel, T., 1999. Guidelines for managing water quality impacts within UK European Marine sites. *Natura 2000 report prepared for the UK Marine SACs Project.* 441 pp., Swindon: Water Research Council on behalf of EN, SNH, CCW, JNCC, SAMS and EHS. [UK Marine SACs Project.], http://www.ukmarinesac.org.uk/

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/

Connor, D.W., Brazier, D.P., Hill, T.O., & Northen, K.O., 1997b. Marine biotope classification for Britain and Ireland. Vol. 1. Littoral biotopes. *Joint Nature Conservation Committee, Peterborough, JNCC Report* no. 229, Version 97.06., *Joint Nature Conservation Committee, Peterborough, JNCC* Report No. 230, Version 97.06.

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.

Crothers, J., 1992. Shell size and shape variation in *Littorina littorea* (L.) from west Somerset. *Proceedings of the Third International Symposium on Littorinid Biology*, J. Grahame, PJ Mill and D. G. Reid (eds.). *The Malacological Society of London*, pp. 91-97.

Crothers, J.H., 1985. Dog-whelks: an introduction to the biology of *Nucella lapillus* (L.) *Field Studies*, **6**, 291-360.

d' Antonia, C., 1985. Epiphytes on the rocky intertidal red alga *Rhodomela latrix* (Turner) C. Agardh: negative effects on the host and food for herbivores? *Journal of Experimental Marine Biology and Ecology*, **86**, 197-218.

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

Davenport, J., Moore, P.G. & LeCompte, E., 1996. Observations on defensive interactions between predatory dogwhelks, *Nucella lapillus* (L.) and mussels, *Mytilus edulis* L. *Journal of Experimental Marine Biology and Ecology*, **206**, 133-147.

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

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

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

Devinny, J. & Volse, L., 1978. Effects of sediments on the development of *Macrocystis pyrifera* gametophytes. *Marine Biology*, **48** (4), 343-348.

Diederich, S., 2005. Differential recruitment of introduced Pacific oysters and native mussels at the North Sea coast: coexistence possible? *Journal of Sea Research*, **53** (4), 269-281.

Dixon, P.S. & Irvine, L.M., 1977. *Seaweeds of the British Isles. Volume 1 Rhodophyta. Part 1 Introduction, Nemaliales, Gigartinales.* London: British Museum (Natural History) London.

Dixon, P.S., 1973. *Biology of the Rhodophyta.* Edinburgh: Oliver & Boyd.

Dommasnes, A., 1969. On the fauna of *Corallina officinalis* L. in western Norway. *Sarsia*, **38**, 71-86.

Duarte, L., Viejo, R.M., Martinez, B., deCastro, M., Gomez-Gesteira, M. & Gallardo, T., 2013. Recent and historical range shifts of two canopy-forming seaweeds in North Spain and the link with trends in sea surface temperature. *Acta Oecologica-International Journal of Ecology*, **51**, 1-10.

Dudgeon, S. R., Davison. L R. & Vadas, R. L.,1989. Effect of freezing on photosynthesis of intertidal macroalgae relative tolerance of *Chondrus crispus* and *Mastocarpus stellatus* (Rhodophyta). *Marine Biology*, **101**, 107-114

Dudgeon, S.R. & Johnson, A.S., 1992. Thick vs. thin: thallus morphology and tissue mechanics influence differential drag and dislodgement of two co-dominant seaweeds . *Journal of Experimental Marine Biology and Ecology*, **165**, 23-43.

Dudgeon, S.R., Davison, I.R. & Vadas, R.L., 1990. Freezing tolerance in the intertidal red algae *Chondrus crispus* and *Mastocarpus stellatus*: relative importance of acclimation and adaptation. *Marine Biology*, **106**, 427-436.

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

Edyvean, R.G.J. & Ford, H., 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.

Edyvean, R.G.J. & Ford, H., 1986. Population structure of *Lithophyllum incrustans* (Philippi) (Corallinales Rhodophyta) from south-west Wales. *Field Studies*, **6**, 397-405.

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

Eriksson, B.K. & Johansson, G., 2003. Sedimentation reduces recruitment success of *Fucus vesiculosus* (Phaeophyceae) in the Baltic Sea. *European Journal of Phycology*, **38** (3), 217-222.

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

Ewers, R., Kasperk, C. & Simmons, B., 1987. Biologishes Knochenimplantat aus Meeresalgen. *Zahnaerztliche Praxis*, **38**, 318-320.

Feare, C.J., 1970b. Aspects of the ecology of an exposed shore population of dogwhelks *Nucella lapillus*. *Oecologia*, **5**, 1-18.

Filion-Myclebust, C. & Norton, T.A., 1981. Epidermis shedding in the brown seaweed *Ascophyllum nodosum* (L.) Le Jolis, and its ecological significance. *Marine Biology Letters*, **2**, 45-51.

Firth, L., Thompson, R., Bohn, K., Abbiati, M., Airoldi, L., Bouma, T., Bozzeda, F., Ceccherelli, V., Colangelo, M. & Evans, A., 2014. Between a rock and a hard place: Environmental and engineering considerations when designing coastal defence structures. *Coastal Engineering*, **87**, 122-135.

Fletcher, R.L. & Callow, M.E., 1992. The settlement, attachment and establishment of marine algal spores. *British Phycological Journal*, **27**, 303-329.

Fletcher, R.L., 1996. The occurrence of 'green tides' - a review. In *Marine Benthic Vegetation. Recent changes and the Effects of Eutrophication* (ed. W. Schramm & P.H. Nienhuis). Berlin Heidelberg: Springer-Verlag. [Ecological Studies, vol. 123].

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.

Fretter, V. & Graham, A., 1962. *British Prosobranch Molluscs.* London: Ray Society.

Fretter, V. & Graham, A., 1964. Reproduction. In *Physiology of Mollusca*, vol. 1, (ed. K.M. Wilbur & C.M. Yonge). New York: Academic Press.

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

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.

Gibbs, P.E., Green, J.C. & Pascoe, P.C., 1999. A massive summer kill of the dog-whelk, *Nucella lapillus*, on the north Cornwall coast in 1995: freak or forerunner? *Journal of the Marine Biological Association of the United Kingdom*, **79**, 103-109.

Glegg, G. A., Hickman, L. & Rowland, S. J., 1999. Contamination of limpets (*Patella vulgata*) following the Sea Empress oil spill. *Marine Pollution Bulletin*, **38**, 119-125.

Green, D., Chapman, M. & Blockley, D., 2012. Ecological consequences of the type of rock used in the construction of artificial boulder-fields. *Ecological Engineering*, **46**, 1-10.

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

Guidone, M. & Grace, S., 2010. The ratio of gametophytes to tetrasporophytes of intertidal *Chondrus crispus* (Gigartinaceae) across a salinity gradient. *Rhodora*, **112** (949), 80-84.

Guiry, M.D. & Blunden, G., 1991. *Seaweed Resources in Europe: Uses and Potential*. Chicester: John Wiley & Sons.

Gutierrez, L.M. & Fernandez, C., 1992. Water motion and morphology in *Chondrus crispus* (Rhodophyta). *Journal of Phycology*, **28**, 156-162.

Hammer, L., 1972. Anaerobiosis in marine algae and marine phanerograms. In *Proceedings of the Seventh International Seaweed Symposium, Sapporo, Japan, August 8-12, 1971* (ed. K. Nisizawa, S. Arasaki, Chihara, M., Hirose, H., Nakamura V., Tsuchiya, Y.), pp. 414-419. Tokyo: Tokyo University Press.

Harlin, M.M., & Lindbergh, J.M., 1977. Selection of substrata by seaweed: optimal surface relief. *Marine Biology*, **40**, 33-40.

Hartnoll, R.G. & Hawkins, S.J., 1985. Patchiness and fluctuations on moderately exposed rocky shores. *Ophelia*, **24**, 53-63.

Hawkins, S.J. & Hartnoll, R.G., 1983. Grazing of intertidal algae by marine invertebrates. *Oceanography and Marine Biology: an Annual Review*, **21**, 195-282.

Hawkins, S.J. & Hartnoll, R.G., 1983b. Changes in a rocky shore community: an evolution of monitoring. *Journal of Marine Environmental Research*, **9**, 195-282.

Hawkins, S.J. & Hartnoll, R.G., 1985. Factors determining the upper limits of intertidal canopy-forming algae. *Marine Ecology Progress Series*, **20**, 265-271.

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

Hawkins, S.J., Hartnoll, R.G., Kain, J.M. & Norton, T.A., 1992. Plant-animal interactions on hard substrata in the north-east Atlantic. In *Plant-animal interactions in the marine benthos* (ed. D.M. John, S.J. Hawkins & J.H. Price), pp. 1-32. Oxford: Clarendon Press. [Systematics Association Special Volume, no. 46.]

Herbert, R.J.H., Roberts, C., Humphreys, J., & Fletcher, S. 2012. The Pacific oyster (*Crassostra gigas*) in the UK: economic, legal and environmental issues associated with its cultivation, wild establishment and exploitation. Available from: http://www.dardni.gov.uk/pacific-oysters-issue-paper.pdf

Hicks, G.R.F., 1985. Meiofauna associated with rocky shore algae. In *The Ecology of Rocky Coasts: essays presented to J.R. Lewis, D.Sc.*, (ed. P.G. Moore & R. Seed, ed.). pp. 36-56. London: Hodder & Stoughton Ltd.

Hill, S., Burrows, S.J. & Hawkins, S.J., 1998. *Intertidal Reef Biotopes (Volume VI). An overview of dynamics and sensitivity characteristics for conservation management of marine Special Areas of Conservation.* Oban: Scottish Association for Marine Science (UK Marine SACs Project)., Scottish Association for Marine Science (UK Marine SACs Project).

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.

Holt, T.J., Hartnoll, R.G. & Hawkins, S.J., 1997. The sensitivity and vulnerability to man-induced change of selected communities: intertidal brown algal shrubs, *Zostera* beds and *Sabellaria spinulosa* reefs. *English Nature, Peterborough, English Nature Research Report* No. 234.

Holt, T.J., Rees, E.I., Hawkins, S.J. & Seed, R., 1998. Biogenic reefs (Volume IX). An overview of dynamic and sensitivity characteristics for conservation management of marine SACs. *Scottish Association for Marine Science (UK Marine SACs Project)*, 174 pp.

Huff, T.M. & Jarett, J.K., 2007. Sand addition alters the invertebrate community of intertidal coralline turf. *Marine Ecology Progress Series*, **345**, 75-82.

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.

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

Irvine, L.M., 1983. *Seaweeds of the British Isles* vol. 1. *Rhodophyta Part 2A. Cryptonemiales (sensu stricto), Palmariales, Rhodymeniales*. London: British Museum (Natural History).

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

Kain, J.M., 1975a. Algal recolonization of some cleared subtidal areas. *Journal of Ecology*, **63**, 739-765.

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.

Keough, M.J. & Quinn, G.P., 1998. Effects of periodic disturbances from trampling on rocky intertidal algal beds. *Ecological Applications*, **8** (1), 141-161.

Kindig, A.C., & Littler, M.M., 1980. Growth and primary productivity of marine macrophytes exposed to domestic sewage effluents. *Marine Environmental Research*, **3**, 81-100.

Kinne, O. (ed.), 1971a. *Marine Ecology: A Comprehensive, Integrated Treatise on Life in Oceans and Coastal Waters. Vol. 1 Environmental Factors, Part 2*. Chichester: John Wiley & Sons.

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. (ed.), 1980. *Diseases of marine animals.* vol. 1. *General aspects. Protozoa to Gastropoda.* Chichester: 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.

Kirby, R.R., Bayne, B.L. & Berry, R.J., 1994b. Physiological variation in the dog-whelk *Nucella lapillus* L., either side of a cline in allozyme and karyotype frequencies. *Biological Journal of the Linnean Society*, **53**, 277-290.

Kitching, J.A., 1987. The fauna and flora associated with *Himanthalia elongata* (L) S. F. Gray in relation to water current and wave action in the Lough Hyne Marine Nature Reserve. *Estuarine, Coastal and Shelf Science*, **25**, 663-676.

Kochmann, J., Buschbaum, C., Volkenborn, N. & Reise, K., 2008. Shift from native mussels to alien oysters: differential effects of ecosystem engineers. *Journal of Experimental Marine Biology and Ecology*, **364** (1), 1-10.

Kraemer, G.P., Sellberg, M., Gordon, A. & Main, J., 2007. Eight-year record of *Hemigrapsus sanguineus* (Asian shore crab) invasion in western Long Island sound estuary. *Northeastern Naturalist*, **14** (2), 207-224.

Kuhlenkamp, R. & Kind, B., 2013. Arrival of the invasive *Watersipora subtorquata* (Bryozoa) at Helgoland (Germany, North Sea) on floating macroalgae (*Himanthalia*). *Marine Biodiversity Records*, **6**, 1-6

Largen, M.J., 1967. The influence of water temperature upon the life of the dog whelk *Thais lapillus* (Gastropoda: Prosobranchia). *Journal of Animal Ecology*, **36**, 207-214.

Latham, H., 2008. Temperature stress-induced bleaching of the coralline alga *Corallina officinalis*: a role for the enzyme bromoperoxidase. *Bioscience Horizons*, 1-10

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.

Lewis, J.R., 1964. *The Ecology of Rocky Shores.* London: English Universities Press.

Lindgren, A. & Åberg, P., 1996. Proportion of life cycle stages of *Chondrus crispus* and its population structure: a comparison between a marine and an estuarine environment. *Botanica Marina*, **39** (1-6), 263-268.

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

Littler, M. & Murray, S., 1975. Impact of sewage on the distribution, abundance and community structure of rocky intertidal macro-organisms. *Marine Biology*, **30** (4), 277-291.

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., & Kauker, B.J., 1984. Heterotrichy and survival strategies in the red alga *Corallina officinalis* L. *Botanica Marina*, **27**, 37-44.

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

Lobban, C.S. & Harrison, P.J., 1997. *Seaweed ecology and physiology*. Cambridge: Cambridge University Press.

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.

Lubchenco, J., 1980. Algal zonation in the New England rocky intertidal community: an experimental analysis. *Ecology*, **61**, 333-344.

Lüning, K., 1990. *Seaweeds: their environment, biogeography, and ecophysiology*: John Wiley & Sons.

Luther, G., 1976. Bewuchsuntersuchungen auf Natursteinsubstraten im Gezeitenbereich des Nordsylter Wattenmeeres: Algen. *Helgoländer Wissenschaftliche Meeresuntersuchungen*, 28 (3-4), 318-351.

MacFarlane, C.I., 1952. A survey of certain seaweeds of commercial importance in southwest Nova Scotia. *Canadian Journal of Botany*, **30**, 78-97.

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

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

Marshall, S., Newton, L. & Orr, A.P., 1949. *A study of certain British seaweeds and their utilization in the preparation of agar*. London: HMSO. 184 pp. *A study of certain British seaweeds and their utilization in the preparation of agar*. London: HMSO. 184 pp., London: HMSO. 184 pp.

Martínez, B., Arenas, F., Trilla, A., Viejo, R.M. & Carreno, F., 2015. Combining physiological threshold knowledge to species distribution models is key to improving forecasts of the future niche for macroalgae. *Global Change Biology*, **21** (4), 1422-1433.

Masaki, T., Fujita, D. & Akioka, H., 1981. Observation on the spore germination of *Laminaria japonica* on *Lithophyllum yessoense* (Rhodophyta, Corallinaceace) in culture. *Bulletin of the Faculty of Fish Research, Hokkaido University*, **32**, 349-356.

Mathieson, A.C. & Burns, R.L., 1971. Ecological studies of economic red algae. 1. Photosynthesis and respiration of *Chondrus crispus* (Stackhouse) and *Gigartina stellata* (Stackhouse) Batters. *Journal of Experimental Marine Biology and Ecology*, **7**, 197-206.

Mathieson, A.C. & Burns, R.L., 1975. Ecological studies of economic red algae. 5. Growth and reproduction of natural and harvested populations of *Chondrus crispus* Stackhouse in New Hampshire. *Journal of Experimental Marine Biology and Ecology*, **17**, 137-156.

May, V., 1985. Observations on algal floras close to two sewerage outlets. *Cunninghamia*, **1**, 385-394.

Van der Meer, J., 1987. Experimental hybridization of *Palmaria palmata* (Rhodophyta) from the northeast and northwest Atlantic Ocean. *Canadian Journal of Botany*, **65** (7), 1451-1458.

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.

Miossec, L., Le Deuff, R.-M. & Goulletquer, P., 2009. Alien species alert: *Crassostrea gigas* (Pacific oyster). *ICES Cooperative Research Report*, 299

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.

Moss, B., 1969. Apical meristems and growth control in *Himanthalia elongata* (SF Gray). *New Phytologist*, **68** (2), 387-397.

Moss, B., Mercer, S., & Sheader, A., 1973. Factors Affecting the Distribution of *Himanthalia elongata* (L.) S.F. Gray on the North-east Coast of England. *Estuarine and Coastal Marine Science*, **1**, 233-243.

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.

Petpiroon, S. & Dicks, B., 1982. Environmental effects (1969 to 1981) of a refinery effluent discharged into Littlewick Bay, Milford Haven. *Field Studies*, **5**, 623-641.

Petraitis, P.S., 1987. Immobilization of the predatory gastropod, *Nucella lapillus*, by its prey *Mytilus edulis*. *Biological Bulletin, Marine Biological Laboratory, Woods Hole*, **172**, 307-314.

Pinn, E.H. & Rodgers, M., 2005. The influence of visitors on intertidal biodiversity. *Journal of the Marine Biological Association of the United Kingdom*, **85** (02), 263-268.

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

Prathep, A. 2001. Population ecology of a turf-forming red alga, *Osmundea pinnatifida* from the Isle of Man, British Isles. Ph.D. thesis. University of Liverpool.

Pratt, M. & Johnson, A. 2002. Strength, drag and dislodgement of two competing intertidal algae from two wave exposures and four seasons. *Journal of Experimental Marine Biology and Ecology*, **272**, 71-101.

Prince, J.S. & Kingsbury, J.M., 1973. The ecology of *Chondrus crispus* at Plymouth, Massachusetts. 3. Effect of elevated temperature on growth and survival. *Biology Bulletin*, **145**, 580-588.

Pringle, J., & Semple, R. 1980. The benthic algal biomass, commercial harvesting, and *Chondrus* growth and colonization off southwestern Nova Scotia. In *Proceedings of the workshop on the relationship between sea urchin grazing and commercial plant / animal harvesting*. Edited by J. Pringle, G. Sharp, and J. Caddy. Canadian Technical Report of Fisheries and Aquatic Sciences 954. pp. 144-169.

Provan, J., Murphy, S. & Maggs, C.A., 2005. Tracking the invasive history of the green alga *Codium fragile* ssp. *tomentosoides*. *Molecular Ecology*, **14**, 189-194.

Raffaelli, D., 1982. Recent ecological research on some European species of *Littorina*. *Journal of Molluscan Studies*, **48** (3), 342-354.

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

Raffaelli, D.G. & Hawkins, S.J., 1996. *Intertidal Ecology* London: Chapman and Hall.

Rai, L., Gaur, J.P. & Kumar, H.D., 1981. Phycology and heavy-metal pollution. *Biological Reviews*, **56**, 99-151.

Rice, H., Leighty, D.A. & McLeod, G.C., 1973. The effects of some trace metals on marine phytoplankton. *CRC Critical Review in Microbiology*, **3**, 27-49.

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.

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.

Russell, G., 1990. Age and stage in seaweed populations: A cautionary tale. *British Phycological Journal*, **25** (3), 245-249.

Russell, G. & Veltkamp, C.J., 1984. Epiphyte survival on skin-shedding macrophytes. *Marine Ecology Progress Series*, **18**, 149-153.

Sandison, E.E., 1968. Respiratory response to temperature and temperature tolerance of some intertidal gastropods. *Journal of Experimental Marine Biology and Ecology*, **1**, 271-281.

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

Schiel, D.R. & Taylor, D.I., 1999. Effects of trampling on a rocky intertidal algal assemblage in southern New Zealand. *Journal of Experimental Marine Biology and Ecology*, **235**, 213-235.

Schmidt, A.L. & Scheibling, R.E., 2007. Effects of native and invasive macroalgal canopies on composition and abundance of mobile benthic macrofauna and turf-forming algae. *Journal of Experimental Marine Biology and Ecology*, **341** (1), 110-130.

Schonbeck, M.W. & Norton, T.A., 1980. Factors controlling the lower limits of fucoid algae on the shore. *Botanica Marina*, **23**, 141-147.

Scrosati, R., Garbary, D.J. & McLachlan, J., 1994. Reproductive ecology of *Chondrus crispus* (Rhodophyta, Gigartinales) from Nova Scotia, Canada. *Botanica Marina*, **37**, 293-300.

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.

Seed, R. & Suchanek, T.H., 1992. Population and community ecology of *Mytilus*. In *The mussel* Mytilus*: ecology, physiology, genetics and culture*, (ed. E.M. Gosling), pp. 87-169. Amsterdam: Elsevier Science Publ. [Developments in Aquaculture and Fisheries Science, no. 25.]

Sewell, J. 2011c. Wireweed, *Sargassum muticum*. *Great Britain Non-native Species Secretariat*. [cited 16/06/2015]. Available from: <http://www.nonnativespecies.org

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.

Sharp, G.J., Tetu, C., Semple, R. & Jones, D., 1993. Recent changes in the seaweed community of western Prince Edward Island: implications for the seaweed industry. *Hydrobiologia*, **260-261**, 291-296.

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

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

Stengel, D.B., Wilkes, R.J. & Guiry, M.D., 1999. Seasonal growth and recruitment of *Himanthalia elongata* (Fucales, Phaeophycota) in different habitats on the Irish west coasts. *European Journal of Phycology*, **34**, 213-221.

Stewart, J.G., 1989. Establishment, persistence and dominance of *Corallina* (Rhodophyta) in algal turf. *Journal of Phycology*, **25** (3), 436-446.

Stickle, W.B., Moore, M.N. & Bayne, B.L., 1985. Effects of temperature, salinity and aerial exposure on predation and lysosomal stability in the dog whelk *Thais* (*Nucella*) *lapillus* (L.). *Journal of Experimental Marine Biology and Ecology*, **93**, 235-258.

Storey, K.B., Lant, B., Anozie, O.O. & Storey, J.M., 2013. Metabolic mechanisms for anoxia tolerance and freezing survival in the intertidal gastropod, *Littorina littorea*. *Comparative Biochemistry and Physiology Part A: Molecular & Integrative Physiology*, **165** (4), 448-459.

Sweet, N.S. 2011j. Green sea-fingers (tomentosoides), *Codium fragile* subsp. *tomentosoides*. *Great Britain Non-native Species Secretariat.* [cited 16/06/2015]. Available from: <http://www.nonnativespecies.org

Sweet, N.S. 2011g. Devil's Tongue Weed, *Grateloupia turuturu* . *Great Britain Non-native Species Secretariat*. [cited 16/06/2015]. Available from: