Fucus serratus and under-boulder fauna on exposed to moderately exposed lower eulittoral boulders

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An underboulder community.
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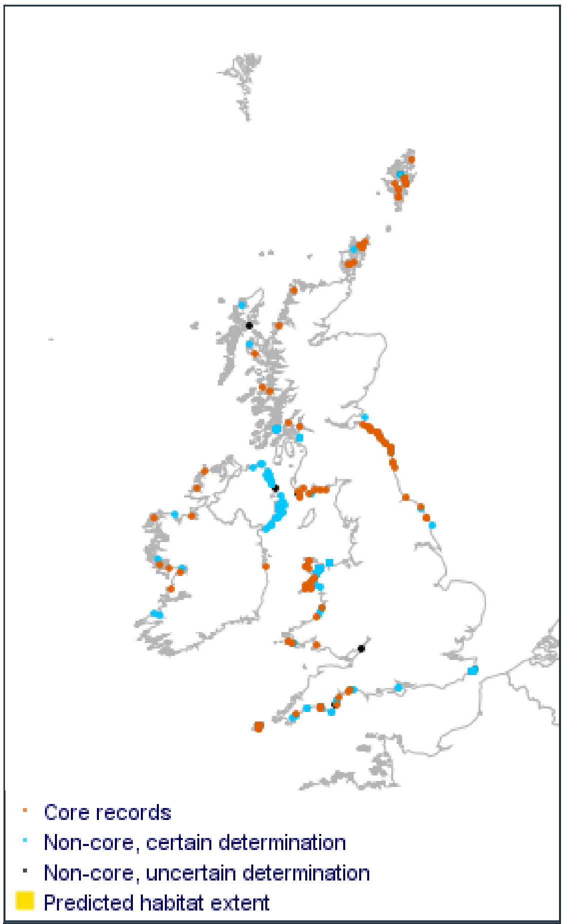

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Biotope distribution data provided by EMODnet Seabed Habitats (www.emodnet-seabedhabitats.eu)

Researched by Dr Heidi Tillin & Frances Perry **Refereed by** Admin

Summary

UK and Ireland classification

EUNIS 2008	A1.2142	*Fucus serratus* and under-boulder fauna on exposed to moderately exposed lower eulittoral boulders
JNCC 2015	LR.MLR.BF.Fser.Bo	*Fucus serratus* and under-boulder fauna on exposed to moderately exposed lower eulittoral boulders
JNCC 2004	LR.MLR.BF.Fser.Bo	*Fucus serratus* and under-boulder fauna on exposed to moderately exposed lower eulittoral boulders
1997 Biotope	LR.MLR.BF.Fser.Fser.Bo	*Fucus serratus* and under-boulder fauna on lower eulittoral boulders

Description

Exposed to moderately exposed lower eulittoral boulders with the wrack *Fucus serratus* community of a high species richness as the presence of the boulders increases the micro-habitat diversity. The upper surfaces of the boulders are colonised by a very similar fauna to the other *Fucus serratus* biotopes, including species such as the limpet *Patella vulgata*, the whelk *Nucella lapillus*, the anemone *Actinia equina* and the barnacle *Semibalanus balanoides*. The shaded sides of the boulders are, depending on environmental conditions, often colonised by a variety of foliose red seaweeds, including *Mastocarpus stellatus*, *Lomentaria articulata*, *Osmundea pinnatifida*, *Palmaria palmata* and *Chondrus crispus*. Coralline algae such as *Corallina officinalis* and coraline crusts, as well as the green seaweeds *Ulva intestinalis* and *Ulva lactuca*, can be found underneath the *Fucus serratus* canopy or in patches on the boulders. The species composition underneath the boulders varies considerably depending on the underlying substratum. On muddy shores the fauna living under the boulders may be limited to a few infaunal species, such as the polychaete *Cirratulus cirratus*. Where more space is available beneath the boulders there may be a rich assemblage of animals. Characteristic mobile species include the crabs *Porcellana platycheles* and *Carcinus maenas*. Also present on and beneath the boulders are the tube-forming polychaete *Spirobranchus triqueter*, spirorbid polychaetes and a few winkles such as *Littorina obtusata/mariae* and *Littorina littorea* or even the top shell *Gibbula cineraria*. Encrusting colonies of the sponge *Halichondria panicea* are also typical of the undersides of boulders, while the hydroid *Dynamena pumila* colonies can be found on the *Fucus serratus* fronds. The richest examples of this biotope also contain a variety of brittlestars, ascidians and small hydroids (JNCC, 2015)

Depth range

Lower shore

Additional information

-

Listed By

- none -

Further information sources

Search on:

G G JNCC

Habitat review

Ecology

Ecological and functional relationships

- The fauna are dominated by a variety of active and passive suspension feeders such as encrusting sponges (e.g. *Halichondria panicea*), solitary ascidians (e.g. *Dendrodoa grossularia*), barnacles (e.g. *Balanus crenatus*), spirorbid worms (e.g. *Spirorbis spirorbis*), hydroids (e.g. *Dynamena pumila*), bryozoans (e.g. the sea mat *Electra pilosa* and the encrusting bryozoan *Umbonula littoralis*) and colonial ascidians (e.g. *Botryllus schlosseri*). There is likely to be competition for space between many of the encrusting species. Gordon (1972) noted that competition between bryozoans and ascidians always favoured ascidians. He also noted that *Halichondria* sponges, even after they had died, prevented the spreading and survival of bryozoans trying to overgrow them.
- Herbivores include the common periwinkle *Littorina littorea*, the grey top shell *Steromphala cineraria*, the green sea urchin *Psammechinus milaris* and, less frequently, the common limpet *Patella vulgata*. Herbivorous grazers most likely use the underboulder habitat for shelter but emerge from under the boulders to feed.
- There are few species that prey on other members of the community but, for instance, dog whelks *Nucella lapillus* may feed on barnacles and the European cowrie *Trivia monacha* feeds on the star ascidian *Botryllus schlosseri*. The common shore crab *Carcinas maenas* is probably the largest mobile predator associated with MLR.Fser.Fser.Bo. It will move between the boulders and pebbles feeding primarily on small molluscs, especially *Littorina* spp., annelids and other crustacea. It will also consume algal material. Non-mobile carnivores include the beadlet anemone *Actinia equina* which feeds passively.
- Some resident mobile species are detritivores such as the hairy porcelain crab *Porcellana platycheles* and some brittle stars.
- Several species that occur under boulders gain shelter from insolation and predators when the tide is out but are not an integral part of the community; for instance, blennies, crabs and shrimps e.g. the common prawn *Palaemon serratus*.

Species diversity and disturbance

Boulder communities are subject to frequent wave-induced disturbance, especially during the winter months as storm and wave energy increases. Due to the varying size of boulders likely to be found in MLR.Fser.Fser.Bo, some boulders will be moved around and turned-over more frequently than others.

Larger boulders remain undisturbed for longer periods of time and, consequently, the community on them is likely to be dominated by a few late successional species. In MLR.Fser.Fser.Bo, large stable boulders may be dominated by a few prolific species such as *Dendrodoa* and *Halichondria* (Foster-Smith, pers. comm.). In contrast, small boulders are tossed around regularly and are unlikely to reach a 'climax' community as disturbance is too frequent. The frequency of disturbance determines the interval of time over which recolonization can occur (Sousa, 1985) and small boulders sample the available pool of spores and larvae more often (Sousa, 1979a) and they are likely to be characterized by hardy species capable of rapidly colonizing bare space e.g. barnacles, spirorbid worms and bryozoans. Sousa (1979a) noted that, in an algal dominated boulder field in California, boulders subjected to intermediate disturbance frequencies were usually less dominated than those which are frequently disturbed, and always less dominated than boulders which were seldom disturbed. Furthermore, intermediate boulders remained undisturbed for long enough that several species had become dominant but not so long that species had been competitively displaced, resulting in dominance. In other words, intermediate size boulders are likely to be more diverse in terms of species diversity. For this reason, the species composition under boulders within the MLR.Fser.Fser.Bo classification can vary considerably which can be problematic when assessing sensitivity (see Species Composition).

Seasonal and longer term change

Some species of bryozoans and hydroids demonstrate seasonal cycles of growth in spring/summer and regression (die back) in late autumn/winter, overwintering as dormant stages or juvenile stages (see Ryland, 1976; Gili & Hughes, 1995; Hayward & Ryland, 1998). Many of the bryozoans and hydroid species are opportunists adapted to rapid growth and reproduction (r-selected), taking advantage of the spring/summer phytoplankton bloom and more favourable (less stormy) conditions (Dyrynda & Ryland, 1982; Gili & Hughes, 1995). Henry (2002) reported a drastic decline in *Dynamena pumila* over the winter months in the Bay of Fundy. Foster-Smith (1989) recorded that many encrusting ascidians increased in abundance by late summer under boulders on the Northumbrian coast.

On the boulder shores with which MLR.Fser.Fser.Bo is associated, the increased storm and wave energy over the winter months are likely to significantly influence both the flora and faunal components of MLR.Fser.Fser.Bo. Many boulders and cobbles will be thrown around creating bare patches in encrusting species, ripping seaweed off the boulders and overturning boulders to the detriment of species previously on top of the boulders which may suffer from anoxia and crushing etc. These species are likely to perish if left under the boulder which will mean that the proportion of 'bare' rock will increase. Over the winter months therefore they may be an increase in opportunistic species such as *Spirobranchus triqueter*. However, the winter months also giver the late successional species a chance to colonize the rocks as other e.g. algae such as *Ulva* die back. Sousa (1979b) found that *Ulva* sp. inhibited the colonization of mid-successional species such as *Fucus serratus* and that these mid-successional species, in turn, inhibited the recruitment of late-successional species such as *Gigartina canaliculata*. Therefore, these winter months are important for the development and diversity of the biotope as a whole.

Habitat structure and complexity

- The epilithic community usually occurs as a single layer although competition between encrusting species may result in overgrowth and smothering.
- Habitat complexity increases where soft rocks are bored by bivalve molluscs creating holes for other species to nestle.
- Variation occurs especially in relation to the degree of influence of underlying sediments. Physical complexity is increased where boulders lie on top of other loosely-packed boulder creating interstices whilst siltation under the boulders means that deeper silty layers may support detritus-feeding polychaetes (Foster-Smith, pers. comm.). Faunal diversity on the boulder surface will be decreased where the boulders are embedded or partly in sediment. In contrast, there may be flowing water under some boulders (for instance, overflows from pools or lagoonal habitats draining at low water) which creates rich communities.

Productivity

Insufficient information

Recruitment processes

The majority of important and characteristic species associated with MLR.Fser.Fser.Bo have planktonic larvae which recruit frequently. Recruitment in the important species is summarized below.

- The breadcrumb sponge *Halichondria panicea* is likely to have a short, annual season of sexual reproduction. Most sponges are hermaphroditic but cross-fertilization normally occurs. The process may be oviparous, where there is a mass spawning of gametes through the osculum which enter a neighbouring individual in the inhalant current. Fertilized eggs are discharged into the sea where they develop into a planula larva. However, in the majority, development is viviparous, whereby the larva develops within the sponge and is then released. Larvae have a short planktonic life of a few hours to a few weeks, so that dispersal is probably limited.
- In *Botryllus schlosseri*, up to eight eggs are produced per zooid. After fertilization and development to a tadpole stage, the tadpole is released and is free swimming for up to 36 hours (Berril, 1950; Berril, 1975). This short planktonic stage therefore limits recruitment to nearby colonies.
- Ingle (1997) indicated that the eggs of *Pisidia longicornis* were present from March to August in southern England and from February to September in the Mediterranean. The planktonic larvae and highly mobile nature of this crab mean that this species does not necessarily rely on recruitment from ,local sources. Underboulder areas may be important refuges for young crabs, especially *Cancer pagurus*.
- The dispersal phase of *Umbonula littoralis* is probably brief and larvae probably do not travel far therefore recruitment is dependant on local sources. Embryos were recorded as present in the Plymouth area in June and August (Marine Biological Association, 1957), from October and November on the north-east coast of England (Hastings, 1944) and from September to February in Manx waters (Eggleston, 1969).
- *Balanus crenatus* releases planktotrophic nauplii larvae between February and September, with peaks in April and late summer when phytoplankton levels are highest. They pass through six nauplii stages before eventually developing into a cyprid larva which is are specialized for settlement. Peak settlement occurs in April and declines until October. The larvae may not settle for a month after release and therefore, dispersal potential is relatively high as is recruitment from distant sources. Metamorphosis usually takes place within 24 hours of settlement.
- Although asexual reproduction occurs in many ascidians, reproduction in the baked bean ascidian *Dendrodoa grossularia* is entirely sexual (Millar, 1954). Millar studied reproduction in *Dendrodoa grossularia* in two locations (the River Crouch in Essex and the Isle of Cumbrae in the Firth of Clyde) and found that reproduction was bi-polar in nature with one peak in spring and another in late autumn, the spring episode being more intense. The average number of eggs produced per individual (over 7 mm in length) was only ca 25-100 eggs in the Clyde and River Crouch respectively. Furthermore, the eggs are brooded internally until the larval stage is reached thereby compressing the free swimming stage.

Time for community to reach maturity

Settlement panels, which attract similar communities to underboulder habitats, may be fully colonized within about 18 months of being placed into the environment (extrapolated from Sutherland & Karlson, 1977; Todd, 1994). Development of 'mature' communities under boulders is likely to occur within two years and there will be dynamic stability, i.e. composition of the community will remain much the same although individual organisms and colonies will die and be replaced by the same species.

Additional information

No text entered

Preferences & Distribution

Habitat preferences

Depth Range	Lower shore
Water clarity preferences	Field Unresearched
Limiting Nutrients	Field unresearched, No preference
Salinity preferences	Full (30-40 psu)
Physiographic preferences	Enclosed coast / Embayment
Biological zone preferences	Lower eulittoral
Substratum/habitat preferences	Large to very large boulders, Small boulders, Under boulders
Tidal strength preferences	No information
Wave exposure preferences	Exposed, Moderately exposed, Sheltered
Other preferences	

Additional Information

Species composition

Species found especially in this biotope

- *Turbicellepora magnicostata*

Rare or scarce species associated with this biotope

- *Turbicellepora magnicostata*

Additional information

Underboulder communities are especially described in Foster-Smith (1989, 1991), Foster-Smith & Foster-Smith (1987), and Hiscock (1984) and in various reports produced during the MNCR survey of Scottish sea lochs.

Sensitivity review

Sensitivity characteristics of the habitat and relevant characteristic species

The biotope description and information on characteristic features is taken from Connor *et al.* (2004). The upper surfaces of the boulders are colonized by a very similar fauna to the other *Fucus serratus* biotopes, including species such as the limpet *Patella vulgata*, the whelk *Nucella lapillus*, the anemone *Actinia equina* and the barnacle *Semibalanus balanoides*. The shaded sides of the boulders are, depending on environmental conditions, often colonied by a variety of foliose red seaweeds, including *Mastocarpus stellatus*, *Lomentaria articulata*, *Osmundea pinnatifida*, *Palmaria palmata* and *Chondrus crispus*. Coralline algae such as *Corallina officinalis* and coralline crusts, as well as the green seaweeds *Ulva intestinalis* and *Ulva lactuca*, can be found underneath the *Fucus serratus* canopy or in patches on the boulders. The characterizing species define the biotope and provide complex habitats that shelter species and modify environmental conditions. The loss of the characterizing species, in particular *Fucus serratus*, could lead to reclassification of the biotope, hence the sensitivity assessments focus on these species.

The species composition underneath the boulders varies considerably depending on the underlying substratum. On muddy shores the fauna living under the boulders may be limited to a few infaunal species, such as the polychaete *Cirratulus cirratus*. Where more space is available beneath the boulders there may be a rich assemblage of animals. Also present on and beneath the boulders are the tube-forming polychaete *Spirobranchus triqueter*, spirorbid polychaetes and a few winkles such *as Littorina obtusata/mariae* and *Littorina littorea* and the top shell *Gibbula cineraria*. Encrusting colonies of the sponge *Halichondria panicea* are also typical of the undersides of boulders, while the hydroid *Dynamena pumila* colonies can be found on the *Fucus serratus* fronds. The sensitivity of these associated species are considered generally.

Depending on the size of boulders, disturbance by wave action may structure this biotope and substratum sensitivity is considered within the assessments where the pressures may alter this element.

Resilience and recovery rates of habitat

No evidence for recovery rates of this specific biotope were found. The algae within the biotope can regrow damaged fronds and blades and may regrow from perennial holdfasts or crustose bases, where these remain. The molluscs and crabs may be able to repair shells following minor damage but will be more vulnerable to predators and desiccation while healing. Where populations of animals and macroalgae are entirely removed (resistance is none) recovery will require recolonization by propagules. Adults of the mobile species present in the biotope, such as limpets and littorinids may recolonize through adult migration into the habitat from adjacent populations following disturbance or via larval recolonization. In general the animals within the biotope, such as *Littorina littorea* and barnacles, produce high numbers of pelagic larvae which are widely distributed by water currents, supporting recolonization from surrounding populations following disturbances. Conversely, the characterizing red and brown macroalgae generally produce eggs which sink rapidly to the substratum in the vicinity of the adult plants and dispersal distances are short (Dudgeon *et al*., 2001). Recovery of algal populations may be rapid where adults remain but prolonged where populations are entirely removed.

Fucus serratus normally lives up to 3 years (Rees, 1932) although in very sheltered areas the plant may live for another couple of years and in very exposed areas may live to only 2 years. High rates of mortality and replacement at all life stages were recorded by Knight & Parke (1950). *Fucus serratus* is dioecious, perennial and reproduces sexually producing high numbers of eggs: Knight & Parke (1950) estimated that large plants produce over a million eggs during the breeding season. Reproduction commences in late spring/early summer, with the proportion of first-year plants reproducing varying by latitude (Knight & Parke, 1950) and continues through summer and autumn, peaking in August - October. Eggs and sperm are released into the water and fertilization occurs in the water column. The zygote then develops into a minute plant that can then settle onto the substratum. Arrontes (1993) determined that the dispersal of *Fucus serratus* gametes and fertilized eggs was restricted to within 1–2 m from the parent. Average annual expansion rates for *Fucus serratus* have been estimated at 0.3 to 0.6 km per year (Coyer *et al*., 2006; Brawley *et al*., 2009). Dispersal is highly limited as the negatively buoyant eggs are fertilized almost immediately after release and dispersal by rafting reproductive individuals is unlikely (Coyer et al., 2006). *Fucus serratus* does not float, and thus mature detached individuals cannot transport reproductive material to distant sites as might be the case for other brown algae. However, *Fucus serratus* is found on all British and Irish coasts so there are few mechanisms isolating populations. While poor dispersal is true for medium or large spatial scales (hundreds of metres to kilometres), recruitment at short distances from parental patches is very efficient, as most propagules settle in the vicinity of parent plants (Arrontes, 2002). Many minute germlings are likely to be present under the parent plants (Knight & Parke, 1950).

Clearance experiments s in the UK using cleared areas of 1 m^2 and clearance of broad strips showed that a population of *Fucus serratus* re-established in one year and increased to almost pre-clearance levels of biomass by the second year (Knight & Parke, 1950). Recruitment in some instances occurred under a canopy of *Ulva* spp., which protected the young plants from wave action (Knight & Parke, 1950). The same results were obtained by Hawkins & Harkin, 1985), who found that after experimental (small scale 2 m^2) canopy removal of *Fucus serratus* on a moderately exposed shore, the *Fucus serratus* cover recovered within one year. Similarly, in a set of clearance experiments (2 x 2m^2 plots) on shores in the Isle of Man, that were dominated by *Ascophyllum nodosum*, Jenkins *et al*. (2004), found that 2 years after clearance *Fucus serratus* had colonized, on average, just under 50% of the cleared plots. Once established, the *Fucus serratus* stands persisted at approximately 50 % mean cover in the cleared areas for the next 10 years of observations, excluding colonization of the plots by *Ascophyllum nodosum*.

The red algae have complex life histories and exhibit distinct morphological stages over the reproductive life history. Alternation occurs between asexual spore producing stages (tetrasporophytes) and male and female plants producing sexually. In habitats where conditions are unfavourable e.g. low salinity habitats, asexual reproduction may maintain populations by retaining genotypes that have evolved to tolerate the prevailing habitat conditions. Life history stages can be morphologically different or very similar. The tetrasporophyte phase of *Mastocarpus stellatus* is known as the petrocelis and is a flat crust, capable of growing laterally and covering extensive areas. The gametophtyes and tetrasporophytes of *Chondrus crispus* are relatively similar; the holdfasts of individual *Chondrus crispus* can coalesce over time and can form an extensive crust on rock (Taylor *et al*., 1981). Other red algae found within the biotope also have life stages that include prostrate creeping bases e.g. encrusting corallines and *Osmundea pinnatifida* whereas in other species such as *Palmaria palmata*, the thallus or fronds arise from a small discoid holdfast. The basal crusts and crustose tetrasporophytes are perennial, tough, resistant stages that may prevent other species from occupying the rock surface and allow rapid regeneration. They may therefore provide a significant recovery mechanism. *Osmundea pinnatifida* turfs may also expand vegetatively where upright fronds bend over, produce rhizoids and colonize adjacent areas of bare rock by acting as stolons (Godin, 1981).

Where holdfasts and basal crusts of red algae are removed, recovery will depend on recolonization via spores. Norton (1992) reviewed dispersal by macroalgae and concluded that dispersal potential is highly variable, recruitment usually occurs on a much more local scale, typically within 10 m of the parent plant. Hence, it is expected that the algal turf would normally rely on recruitment from local individuals and that recovery of populations via spore settlement, where adults are removed, could be protracted. Minchinton *et al.* (1997) documented the recovery of *Chondrus crispus* after a rocky shore in Nova Scotia, Canada, was totally denuded by an ice scouring event. Initial recolonization was dominated by diatoms and ephemeral macroalgae, followed by fucoids and then perennial red seaweeds. After 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. Other red algal species may colonize and spread more slowly. Observed expansion rates of *Osmundea pinnatifida* turfs are very low: 0.015 mm/month in limpet exclusion areas and 0.003 mm/month with limpets (Prathep, 2001).

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 3 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.

Coralline crust is a generic term that in UK biotopes refers to nongeniculate (crustose) species from the family Corallinacea that could include *Lithophyllum incrustans* which is noted to form thick crusts in tidepools, especially in the south west (Adey & Adey, 1973), Although ubiquitous in marine coastal systems, little is understood about the taxonomy, biology and ecology of this taxa (Littler & Littler, 2013). Throughout the sensitivity assessments the term coralline crust is used to refer to the Corallinacea that occur within the biotope. Due to the lack of evidence for species the assessments are generic, although species specific information is presented where available. 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) on 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 spore abundance varies seasonally. Spore survival is extremely low with only a tiny proportion of spores eventually recruiting to the adult population (Edyvean & Ford, 1986). Edyvean & Ford (1984a) 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, 1984a). 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 3 mm/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.

The green algae associated with this biotope are classified as opportunistic species that are able to rapidly colonize newly created gaps across a range of sediment types, shore heights, wave exposures and salinity regimes. *Ulva* sp. release zoospores and gametes (collectively called swarmers) to the water column in high numbers during extended reproduction periods (Smith, 1947). Swarmers are capable of dispersal over a considerable distance, for instance, Amsler & Searles (1980) showed that swarmers of a coastal population of *Ulva* (as *Enteromorpha*) reached exposed artificial substrata on a submarine plateau 35 km away. *Ulva intestinalis* is amongst the first multicellular algae to appear on substrata that have been cleared following a disturbance: following the *Torrey Canyon* oil spill in March 1967, species of the genus *Ulva* rapidly recruited to areas where oil had killed the herbivores that usually grazed on them, so that a rapid greening of the rocks (owing to a thick coating of *Ulva* spp.) was apparent by mid-May (Smith, 1968).

Other species that are associated with this biotope, including the limpet *Patella vulgata*, the barnacle *Semibalanus balanoides* and littorinds generally have slower recovery rates due to episodic recruitment and slower growth. Where individuals are removed from a small area, adult limpets and *Littorina saxatilis* may recolonize from surrounding patches of habitat where these are present. The barnacles and limpets and the winkle *Littorina littorea* are common, widespread species that spawn annually producing pelagic larvae that can disperse over long distances. It is therefore likely that adjacent populations will provide high numbers of larvae, although recruitment may be low due to habitat unsuitability and the presence of algae preventing settlement on rock surfaces. *Littorina saxatilis* brood young and do not have a pelagic life stage, recovery will therefore depend on the presence of adults in close proximity to impacted areas. *Gibbula cineraria*, another grazer within the biotope, is a fast growing species with a short-lifespan (Schöne *et al*., 2007) and pelagic dispersal stages (Underwood, 1972) and is considered to recover quickly (resilience is ‘High’ through migration from adjacent habitats and larval recolonization from any level of impact).

The larvae of the sea squirt *Ascidiella aspersa* have a short free-swimming planktonic stage. Fertilization to settlement and metamorphosis is estimated to only take about 24 hours at 20 °C (Niermann-Kerkenberg & Hofmann, 1989). The sea squirt *Ascidiella scabra* has a high fecundity and settles readily, probably for an extended period from spring to autumn. Svane (1988) describes it as "an annual ascidian" and demonstrated recruitment onto artificial and scraped natural substrata. It is also likely that *Ascidiella scabra* larvae are attracted by existing populations and settle near to adults (Svane *et al*., 1987). Fast growth means that a dense cover could be established within about 2 months. However, if mortality occurs at a time when larvae are not being produced, other species may settle and dominate in the freed spaces. The settlement of new colonies of the breadcrumb sponge *Halichondria panicea* is likely to occur within one year with growth rate ranging from 0.1 to 0.4 cm^2/day. Knowlton & Highsmith (2005) found a rapid response to tissue damage from nudibranch grazing with the sponge recovering within 4 weeks from grazing impacts.

It is likely that habitats where this biotope occurs may undergo much temporal variation mediated by disturbance (particularly where boulders are mobile), competition for space and grazing. The grazers are a key component of this biotope and will influence the dynamics of the biotope. Changes and recovery trajectories following the removal of species are unpredictable and interactions between the characterizing and associated species may be positive or negative. Due to species interactions, recovery trajectories cannot be predicted by life history characteristics alone.

Red algae that form turfs, especially *Corallina officinalis*, are often highly resilient to disturbance, and can recover and reach greater abundance compared to prior disturbance conditions (Bulleri *et al*., 2002; Bertocci *et al*., 2010). Turf algae can then prevent recovery of fucoids and other species by inhibiting recruitment. Mrowicki *et al*. (2014) found that limpet and barnacle removal allowed ephemeral and fucoid macroalgae to establish on sheltered and wave exposed shores in Ireland. Experimental studies have shown that limpets and other grazers control the development of macroalgae by consuming microscopic phases (Jenkins *et al*., 2005) or the adult stages (Davies *et al*., 2007) and can therefore structure biotopes through feeding preferences (Underwood, 1980; Hawkins & Hartnoll, 1985) 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. A change in the abundance of *Patella vulgata* or other grazers could therefore prevent or alter the recovery of this biotope. Opportunistic ephemeral green algae such as *Ulva* sp. can rapidly colonize gaps. These green ephemeral algae are major competitors of *Fucus serratus* for space colonization and nutrient uptake. Blooms of ephemeral algae facilitated by disturbance, particularly where grazers are removed may then slow the development of longer-lived perennial algae, especially fucoids. On the wave exposed and scoured shores that this biotope occurs on, grazing may limit initial settlement of macroalgae but wave action will limit the presence of adults and larger species through breakage and drag effects leading to loss. Limpets and littorinids may enhance barnacle settlement by grazing and removing algae (Hawkins, 1983) or by depositing pedal mucus trails that attract barnacle larvae (Holmes *et al*., 2005). Barnacles may enhance survival of small limpets by moderating environmental stresses but they may also have negative effects on recruitment by occupying space and by limiting access to grazing areas (Lewis & Bowman,1975).

Resilience assessment. Where small patches of *Fucus serratus* are impacted, or where disturbance removes the upper or lower margins of the bed, adjacent adults will supply propagules for recolonization. If the entire population of *Fucus serratus* is lost, other species may come to dominate. Where resistance is 'Medium', loss of (<25% of populations of key species), resilience of *Fucus serratus* and red algal turfs is assessed as 'High', based on rapid regrowth and recolonization due to efficient fertilization rates and recruitment over short distances. Removal of some of the adult canopy will allow the understorey germlings to grow faster. However, if the population of *Fucus serratus* is removed (resistance is 'None'), recovery may take longer, perhaps up to 10 years so the resilience would be scored as 'Medium'. Recovery rates of red algae will be greatly influenced by whether the crust or holdfasts 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 from ice scour (Minchinton *et al*., 1997) recovery is assessed as ‘Medium’ (2-10 years). As recovery, where turfs are removed over large areas, 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 species are removed should 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 or overestimate recovery time, which will be influenced by pressure and site-specific factors. If specimens of *Fucus serratus* remain in small quantities it is likely that re-growth will occur rapidly due to efficient fertilization rates and recruitment over short distances. Recovery is likely to occur within two years resulting in a ‘High’ resilience score. However, if the population is removed (resistance is ‘None’), recovery may take longer, perhaps up to 10 years so the resilience would be scored as ‘Medium’.

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

Hydrological Pressures

	Resistance	Resilience	Sensitivity
Temperature increase (local)	High Q: High A: High C: High	High Q: High A: High C: High	Not sensitive Q: High A: High 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, 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). Local populations may be acclimated to the prevailing temperature regime and may, therefore, exhibit different tolerances to other populations subject to different conditions and therefore caution should be used when inferring tolerances from populations in different regions.

Fucus serratus is found along the Atlantic coast of Europe from Svalbard to Portugal and on the shores of north-east America. The seaweed is thus well within its thermal range in the British

Isles. Nielsen *et al.* (2014) found no negative effects on growth rates of adult *Fucus serratus* to water temperatures of 22°C (based on a laboratory experiment with specimen collected from Firth of Forth, Scotland) and Arrontes (1993) observed that *Fucus serratus* survived in laboratory experiments for 1 week at 25°C. Nielsen *et al.* (2014) did, however, report that germlings were negatively affected by increased temperature indicating that early life stages are more vulnerable than mature algae to this pressure.

Several studies have observed adverse effects of *Fucus serratus* as a result of warm thermal stress in terms of growth, physiological performance and reproductive output in Spain and Portugal (Pearson *et al.*, 2009; Viejo *et al.*, 2011; Martínez *et al.*, 2012). Jueterbock *et al.* (2014) determined that these negative impacts can be explained by restricted within-population genetic diversity. South west Ireland and Brittany are hot-spots of genetic diversity (Coyer *et al.*, 2003; Hoarau *et al.*, 2007) and may thus be more resilient to changes in temperature. Phenotypic plasticity, therefore, plays an important role in determining the sensitivity of individual populations to changes in temperature.

Littler & Kauker (1984) suggested that the crustose bases of *Corallina officinalis* are more resistant of desiccation or heating than fronds. 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 , 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. Fronds of *Osmundea pinnatifida* bleach and die-back in summer, while the crustose holdfasts remain, suggesting that, as for *Corallina officinalis*, the bases are more resistant. The factors responsible for the die-back are a combination of temperature (Flores-Maya, *et al.*, 1992, abstract only) desiccation and irradiance (Prathep, 2001).

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* observed around the edges of pools due to desiccation. Occasional damaged specimens of *Palmaria palmata*, *Osmundea pinnatifida* and *Mastocarpus stellatus* were observed. *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). Temperatures at or above 15 °C may induce physiological stress (Werner & Dring, 2011; Morgan *et al.*, 1980). In tank cultures of *Palmaria palmata* at 20°C and above, all plants were dead within a week (Morgan *et al.*, 1980). Populations may be acclimated to typical conditions but it is likely that *Palmaria palmata* may be bleached or damaged by higher than usual temperatures.

Ulva spp. are characteristic of upper shore rock pools, where water and air temperatures are greatly elevated on hot days. Empirical evidence for thermal tolerance to anthropogenic increases in temperature is provided by the effects of heated effluents on rocky shore communities in Maine, USA. *Ascophyllum* and *Fucus* were eliminated from a rocky shore heated to 27-30°C by a power station whilst *Ulva intestinalis* (as *Enteromorpha intestinalis*) increased significantly near the outfall (Vadas *et al*., 1976).

Limpets, *Patella vulgata* and littorinids also 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 °C (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). *Semibalanus balanoides* and *Patella vulgata* are 'northern' with their range extending from Portugal or northern Spain to the Arctic circle. Populations in the southern part of England are therefore relatively close to the southern edge of their geographic range. Reproductive and recruitment success in both species is linked to temperature and long-term changes in temperature (exceeding the duration of the pressure benchmark) may lead to replacement by the warm water species *Chthamalus montagui* and *Chthamalus stellatus* (Southward *et al*., 1995). In northern Portugal warming seas appear to be linked to a shortening of the reproductive period and the lack of multiple spawning events in *Patella vulgata* and other northern species (Ribeiro *et al*., 2009). Increases in temperature above 12°C induce spawning in *Gibbula cineraria* (Underwood, 1972; Clare, 1990). Individuals from a population from north-east England spawned between late June and early September although females (but not males) could be induced to spawn in winter in response to a temperature increase but males could not (Clare, 1990). An acute increase in temperature that induced spawning in females but not males or that resulted in spawning and fertilization when other conditions were unsuitable would reduce recruitment success (Clare, 1990). This effect, at the duration of the pressure benchmark, is considered to be sub-lethal to the adult population.

Sensitivity assessment. *Fucus serratus* are found in the middle of their natural temperature range in the British Isles and are therefore not likely to be affected by an increase in temperature at the pressure benchmark. An increase in acute or chronic temperature above average British and Irish temperatures is not likely to have a detrimental effect of *Fucus serratus* and associated communities, based on global distribution. However, it should be noted that phenotypic plasticity will influence the tolerance of individual population. Some of the understorey of red algae, such as *Palmaria palmata* may be lost during acute temperature increases if these occur in the summer when plants are already close to the limit of thermal tolerances. 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 may lead to mortality. Sensitivity of *Patella vulgata* and *Semibalanus balanoides* to longer-term, broad-scale perturbations would potentially be greater due to effects on reproduction.

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

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

Lüning (1984) reported that *Fucus serratus* survived in the laboratory for a week a range temperature between 0°C and 25°C. *Fucus serratus* is found along the Atlantic coast of Europe from Svalbard to Portugal and on the shores of north-east America. The seaweed is, thus, well within its thermal range in the British Isles.

No evidence was found for tolerance of decreased temperatures by *Osmundea pinnatifida*. The red algae that characterize this biotope are likely to be tolerant.. 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. The associated species *Mastocarpus stellatus* has a broad geographical distribution (Guiry & Guiry, 2015) and throughout its range experiences wide variation in temperatures (although local populations may be acclimated to the prevailing thermal regime). The photosynthetic rate of *Mastocarpus stellatus* higher on the shore fully recovered from 24 hrs at -20°C (Dudgeon *et al*. (1989). Photosynthesis in *Mastocarpus stellatus* also recovered quickly after experimental freezing (Dudgeon *et al*., 1989, 1995). *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. In the laboratory, plants only became fertile if left at temperatures between 5-7°C with a short light period (Van der Meer, 1979). Acute or chronic changes in temperature below 5°C may therefore reduce reproductive success although reproduction and vegetative growth in warmer months should compensate for any reduction in output. The green algae, *Ulva* spp. 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 cold tolerance of *Semibalanus balanoides* collected in the winter (and thus acclimated to lower 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 found within the turf may occur down the shore to a zone where exposure time to the air (and hence time in freezing temperatures) is less.

Sensitivity assessment. A decrease in acute or chronic temperature above average British and Irish temperatures is not likely to have a detrimental effect of *Fucus serratus* and associated communities, based on global distribution. However, it should be noted that phenotypic plasticity will influence the tolerance of individual population. 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: Medium	Medium Q: High A: Low C: Medium	Medium Q: High A: Low C: Medium

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

Fucoids are able to compensate for changes in salinity by adjusting internal ion concentrations. However this will occur at a cost, reducing photosynthetic rate and hence affecting the growth rate of the seaweed. *Fucus serratus*, commonly inhabit narrow fjords where salinity can vary widely along a spatial (kms) and/or temporal (hours to daily) scale. Growth rates for *Fucus serratus* are maximal at a salinity of 20 psu with the critical limit for recruitment set at 7 psu (Malm *et al*., 2001). An increase in salinity at the pressure benchmark could therefore impact growth.

The associated species are typically found in a range of salinities and tolerance of the red algae varied between species. *Corallina officinalis* is found in tide pools where salinities may fluctuate markedly during exposure to the air. Kinne (1971b) cites maximal growth rates for *Corallina officinalis* between 33 and 38 psu in Texan lagoons. *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).

No evidence was found to assess the salinity tolerances of *Osmundea pinnatifida*.

The associated *Ulva* species can survive hypersaline conditions in supralittoral rockpools subjected to evaporation and are considered to be a very euryhaline, tolerant of extreme salinities ranging from 0 psu to 136 psu (Reed & Russell, 1979). Some variations in salinity tolerance between populations of *Ulva intestinalis* have been found, however, suggesting that plants have some adaptation to the local salinity regime (Alströem-Rapaport *et al*., 2010; Reed & Russell (1979). Increased salinity is most likely to occur in the region of the littoral fringe and supralittoral zone and specimens from these areas were able to tolerate very high salinities, a significant decrease in regeneration only being recorded after exposure to concentrated seawater (102 psu and 136 psu) for > 7 days (Reed & Russell, 1979).

In the laboratory, *Semibalanus balanoides* was found to tolerate salinities between 12 and 50 psu

(Foster, 1970). Young *Littorina littorea* inhabit rock pools where salinity may increase above 35 psu. Thus, the associated species may be able to tolerate some increase in salinity.

Sensitivity assessment. Little direct evidence was found to assess sensitivity to this pressure. Although some increases in salinity may be tolerated by the associated species present, these are generally short-term and mitigated during tidal inundation. This biotope is considered, based on distribution of *Fucus serratus* and the associated red algal species on the mid to 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'.

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

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 at the pressure benchmark a change to variable (18-35 ppt) or reduced (18-30 ppt) is considered. Biotopes found in the intertidal will naturally experience fluctuations in salinity where evaporation increases salinity and inputs of rainwater expose individuals to freshwater. Species found in the intertidal are therefore likely to have some form of behavioural or physiological adaptations to changes in salinity.

Fucoids are able to compensate for changes in salinity by adjusting internal ion concentrations. However this will occur at a cost, reducing photosynthetic rate and hence affecting the growth rate of the seaweed. Growth rates for *Fucus serratus* are maximal at a salinity of 20 psu with the critical limit for recruitment set at 7 psu (Malm *et al*., 2001). Sufficient salinity is essential for successful fertilization and germination in *Fucus* (e.g., Brawley, 1992a; Serrão *et al*., 1999). Malm *et al*. (2001) found that fertilization success in *Fucus serratus* decreased substantially with strongly reduced salinity. Indeed the study found that fertilisation success was 87% at 9 psu but declined to 5% at 6 psu (Malm *et al*., 2001). Reduced salinity does also affect dispersal by decreasing swimming performance of fucoid sperm (Serrão *et al*., 1996a).

No evidence was found to assess the salinity tolerances of *Osmundea pinnatifida*.

Corallina officinalis is found in tide pools 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. *Ulva* species can survive hypersaline conditions in supralittoral rockpools subjected to evaporation and is considered to be a very euryhaline species, tolerant of extreme salinities ranging from 0 psu to 136 psu (Reed & Russell, 1979).

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).

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). In Danish waters where salinities are lowered by freshwater inputs, cultivation experiments found that *Palmaria palmata* growth over a seven month period was greatest at a site where mean salinity was 21±3 psu, while no growth occurred at a site where mean salinity was 17±5 psu (Bak, 2014).

A decrease in salinity may lead to replacement of more sensitive red algal turf forming species by those more tolerant of the changed conditions. *Chondrus crispus* occurs in areas of 'low' salinity. For example, the species occurs in estuaries in New Hampshire, USA, where surface water salinity varies from 16-32 psu (Mathieson & Burns, 1975). Mathieson & Burns (1971) recorded maximum photosynthesis of *Chondrus crispus* in culture at 24 psu, but rates were comparable at 8, 16 and 32 psu. Tasende & Fraga (1999) cultured *Chondrus crispus* spores from north west Spain and concluded that growth was correlated with salinity between 23 and 33 psu.

The associated *Ulva* species can survive hypersaline conditions in supralittoral rockpools subjected to evaporation and are considered to be a very euryhaline, tolerant of extreme salinities ranging from 0 psu to 136 psu (Reed & Russell, 1979). Some variations in salinity tolerance between populations of *Ulva intestinalis* have been found, however, suggesting that plants have some adaptation to the local salinity regime (Alströem-Rapaport *et al*., 2010; Reed & Russell (1979). Increased salinity is most likely to occur in the region of the littoral fringe and supralittoral zone and specimens from these areas were able to tolerate very high salinities, a significant decrease in regeneration only being recorded after exposure to concentrated seawater (102 psu and 136 psu) for >7 days (Reed & Russell, 1979).

Semibalanus balanoides are tolerant of a wide range of salinity and can survive periodic emersion in freshwater, e.g. from rainfall or freshwater run-off, by closing their opercular valves (Foster, 1971b). They can also withstand large changes in salinity over moderately long periods of time by falling into a ‘salt sleep’ and can be found on shores (example from Sweden) with large fluctuations in salinity around a mean of 24 (Jenkins *et al*., 2001). In areas of permanently reduced salinity the Australian barnacle *Austrominius* (formerly *Elminius*) *modestus* may be favoured, as this species is more tolerant of lower salinities), although this is balanced against its lower tolerance of wave exposure. *Littorina littorea* is found in waters of full, variable and reduced salinities (Connor *et al*., 2004) and so populations are not likely to be highly intolerant of decreases in salinity. Therefore, it appears that the biotope would have low intolerance to a decrease in salinity. On return to normal conditions recovery is likely to be very rapid. *Patella vulgata* can tolerate varying salinities and its distribution extends into the mouths of estuaries surviving in salinities down to about 20 psu. However, growth and reproduction may be impaired in reduced salinity. Little *et al.* (1991), for example, observed reduced levels of activity in limpets after heavy rainfall and in the laboratory activity completely stopped at 12 psu. The species can endure periods of low salinity and was found to die only when the salinity was reduced to 3-1 psu (Fretter & Graham, 1994). In experiments where freshwater was trickled over the shell, Arnold (1957) observed limpets withdrawing and clamping the shell onto the substratum. There appears to be an increasing tolerance of low salinities from the lower to the upper limit of distribution of the species on the shore (Fretter & Graham, 1994).

Sensitivity assessment. Most of the literature found on this topic considered short-term (days to weeks) impacts of changes to salinity whilst the benchmark refers to a change for one year. A reduction in salinity at the level benchmark (e.g. from 'Full' to 'Reduced' for one year) could have beneficial effects *on Fucus serratus* as growth rates are maximal below full saline conditions. Prolonged reduction in salinity, e.g. from full to reduced or variable, is likely to reduce the species richness of the biotope due to loss of less tolerant red algae and some intolerant invertebrates. The sensitivity of *Palmaria palmata* for example, appears to be greater than some other turf-forming species such as *Chondrus crispus*. A change in salinity at the pressure benchmark is considered to lead to some mortality of *Palmaria palmata* although the tolerance threshold reported in laboratory and cultivation studies is slightly lower than the assessed benchmark. A change in salinity at the pressure benchmark from full to variable would lead to biotope reclassification to the variable salinity biotope LR.LLR.FVS.FserVS. In that biotope *Fucus serratus* competes with *Fucus vesiculosus* and *Ascophyllum nodosum* (JNCC, 2015). Due to the variable or low salinity conditions the individual red seaweeds may not be as large as specimens found in fully marine conditions and they can lack sexually reproductive structures (JNCC, 2015). Based on changes in dominant species and reduction in habitat suitability for the characterizing species, biotope resistance is assessed as 'Low' and resilience as 'Medium' following habitat restoration. Biotope sensitivity is therefore 'Medium'.

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

Moderate water movement is beneficial to seaweeds as it carries a supply of nutrients and gases to the plants and removes waste products. Propagule dispersal, fertilization, settlement, and recruitment are also influenced by water movement (Pearson & Brawley, 1996). Increased water flow can increase scour through increased sediment movement. Small life stages of macroalgae are likely to be affected by removal of new recruits from the substratum reducing successful recruitment (Devinny & Volse, 1978) (see 'siltation' pressures). A reduction in water flow can cause a thicker boundary layer resulting in lower absorption of nutrients and CO_2 by the macroalgae. Slower water movement can also cause oxygen deficiency directly impacting the fitness of algae (Olsenz, 2011). Higher water flow rates increase mechanical stress on macroalgae by increasing drag. This can result in individuals being torn off the substratum. Jonsson *et al.* (2006) found that flow speed of 7-8 m/s completely dislodged *Fucus vesiculosus* individuals larger than 10 cm. Smaller individuals are likely to better withstand increased water flow as they experience less drag. The risk of dislodgement is greater where algae are attached to pebbles instead of bedrock (Isaeus, 2004). As water velocity increases, algae 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 (Denny *et al*., 1998; Boller & Carrington, 2007). These characteristics allow these species to persist on shores that experience a range of flow speeds. For example, *Mastocarpus stellatus* occurs 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. The habitat structure created by canopies and turfs reduce the effects of water flows on individuals by slowing and disrupting flow (Boller & Carrington, 2006) although this effect will be reduced in this biotope where *Fucus serratus* and red algae occur as scattered plants. The coralline crusts characterizing this biotope are securely attached and as these are flat are subject to little or no drag.

No direct evidence was found to assess the sensitivity of *Osmundea pinnatifida* to changes in water flow at the pressure benchmark. Biogenic habitat structures reduce the effects of water flows on individuals by slowing and disrupting flow. The fronds of *Osmundea pinnatifida* where these are dense will reduce the flow experienced by the turf. **Boller and Carrington (2006**), for example, found that the canopy created by the taller turf of *Chondrus cripsus* reduced drag forces on individual plants by 15-65%.

The crustose holdfasts of *Osmundea pinnatifida*, *Corallina officinalis*, the coralline crusts and the *Mastocarpus stellatus* petrocelis stage are securely attached and as these are relatively flat, are subject less drag than upright fronds and are likely to tolerate changes in water flows at the pressure benchmark. 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 dislodged.

Growth and reproduction of *Semibalanus balanoides* is influenced by food supply and water velocity (Bertness *et al*., 1991). Laboratory experiments demonstrate that barnacle feeding behaviour alters over different flow rates but that barnacles can feed at a variety of flow speeds (Sanford *et al*., 1994). Flow tank experiments using velocities of 0.03, 0.07 and 0.2 m/s showed that a higher proportion of barnacles fed at higher flow rates (Sanford *et al*., 1994). Feeding was passive, meaning the cirri were held out to the flow to catch particles; although active beating of the cirri to generate feeding currents occurs in still water (Crisp & Southward, 1961). Field observations at sites in southern New England (USA) that experience a number of different measured flow speeds, found that *Semibalanus balanoides* from all sites responded quickly to higher flow speeds, with a higher proportion of individuals feeding when current speeds were higher. Barnacles were present at a range of sites, varying from sheltered sites with lower flow rates (maximum observed flow rates <0.06- 0.1 m/s), a bay site with higher flow rates (maximum observed flows 0.2-0.3 m/s) and open coast sites (maximum observed flows 0.2-0.4 m/s). Recruitment was higher at the site with flow rates of 0.2-0.3 m/s (although this may be influenced by supply) and at higher flow microhabitats within all sites. Both laboratory and field observations indicate that flow is an important factor with effects on feeding, growth and recruitment in *Semibalanus balanoides* (Sanford *et al*., 1994, Leonard *et al*., 1998), however, the results suggest that flow is not a limiting factor determining the overall distribution of barnacles as they can adapt to a variety of flow speeds.

Patella vulgata 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, 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, where *Carcinus maenas* is more prevalent, shell apertures are small to inhibit predation. 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).

Gibbula cineraria also appear to tolerate a range of wave exposures from exposed sites to those that are very sheltered (Frid & Fordham, 1994). As with *Littorina littorea*, the morphology of the shell varies according to wave exposure, allowing individuals to adapt to different conditions in the habitat in which the larvae settle (Frid & Fordham, 1994). *Gibbula cineraria* is, however, absent from areas with very strong and turbulent flow

Sensitivity assessment. Based on the available evidence the characterizing species and associated macroalgae and animals are able to occur within a range of flow flow rates and the biotope is therefore considered to be 'Not sensitive' to an increase in water flow at the pressure benchmark. A decrease in water flow may have some effects on recruitment and growth but this is not considered to be lethal at the pressure benchmark and resistance is therefore assessed as 'High' and resilience as 'High' by default, so that the biotope is considered to be 'Not sensitive'. Changes in water flow at the pressure benchmark may result in increased or decreased sediment deposition, these are not considered to alter the character of the biotope which is characterized by sediments beneath boulders (Connor *et al*., 2004).

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

Emergence regime is a key factor structuring intertidal biotopes. Changes in emergence can lead to: greater exposure to desiccation, temperature and salinity variation, reduced levels of time for filter feeding and nutrient uptake and photosynthesising opportunities for the characterizing species. Changes in emergence can also alter competitive interactions and trophic interactions such as grazing and predation. This biotope occurs in the lower shore (Connor *et al*., 2004). This biotope usually occurs immediately below a *Fucus vesiculosus*-barnacle mosaic (FvesB) on moderately exposed shores or a dense canopy of *Fucus vesiculosus* (Fves) or *Ascophyllum nodosum* (Asc.FS) on sheltered shores. The sublittoral fringe below on moderately exposed shores is dominated by the kelp *Laminaria digitata* (Ldig; Ldig.LdigBo), while the kelp *Laminaria saccharina* may co-dominate on sheltered shores (Slat.Ldig; Slat.Ft). Low abundance of these kelp may also occur in this biotope (Connor *et al*., 2004).

Environmental factors partly set upper and lower limits of algal distribution on shores. Spores and developing germlings are particularly susceptible to desiccation as they have very large surface-to-volume ratios, although they benefit from the film of water that persists in concavities on the substratum (Kain & Norton, 1990). At higher shore levels red algae tend to occur only under canopy forming species, as these limit exposure to desiccation (Hawkins & Hartnoll, 1983).

Fucus serratus is more susceptible to desiccation than other *Fucus* species that are located further up the shore and subjected more frequently to aerial exposure (Schonbeck & Norton, 1978). In experiments by Schonbeck & Norton (1978), *Fucus serratus* did not survive transplantation further up the shore, e.g. in the *Fucus spiralis* belt. The critical water content for *Fucus serratus* is estimated at 40% with water losses past this point causing irreversible damage. Beer *et al*. (2014) found that *Fucus serratus* could not regain any positive photosynthetic rates after rehydrating from 10% water content. The upper shore extent of *Fucus serratus* populations may be replaced by species more tolerant of desiccation and more characteristic of the mid-eulittoral such as *Fucus vesiculosus* or *Ascophyllum nodosum*.

The red algae within the biotope are likely to be sensitive to increased emergence. *Corallina officinalis* are sensitive to desiccation (Dommasnes, 1969) and are generally not found on open rock unless protected by algal canopies or where the surfaces are damp or wet. At Hinkley Point (Somerset, England), for example, seawater run-off from deep pools high in the intertidal supports dense turfs of *Corallina* spp. lower on the shore (Bamber & Irving, 1993). Fronds are highly intolerant of desiccation and do not recover from a 15% water loss, which might occur within 40-45 minutes during a spring tide in summer (Wiedemann, 1994). Bleached corallines were observed 15 months after the 1964 Alaska earthquake which elevated areas in Prince William Sound by 10 m. Similarly, increased exposure to air caused by upward movement of 15 cm due to nuclear tests at Armchitka Island, Alaska, adversely affected *Corallina pilulifera* (Johansen, 1974). During an unusually hot summer, Hawkins & Hartnoll (1985) observed damaged *Corallina officinalis* and other red algae. Littler & Kauker, (1984) suggest that the basal crustose stage is adaptive, allowing individuals to survive periods of physical stress as well as physiological stress such as desiccation and heating. The basal crust stage may persist for extended periods with frond regrowth occurring when conditions are favourable.

Osmundea pinnatifida turfs growing on the upper extent of its usual zone in the Isle of Man experience greater desiccation and are shorter and less dense than those lower on the shore (Prathep, 2001), suggesting that habitat quality (measured through growth) decreases with increasing shore height. In laboratory experiments short-term photosynthesis of *Osmundea pinnatifida* was inhibited where fronds had lost more than 50% of their water content (Prathep, 2001). Following resubmergence fronds that had lost 50% of water content had fully recovered (measured as photosynthesis reaching maximal value) after 1 hour while fronds exposed to 70% water loss took 5 hours to recover (Prathep, 2001). Repeated exposure to high levels of desiccation would clearly impact growth.

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 also limits the upper shore extent of this biotope (Hawkins & Hartnoll, 1985). These observations and further grazer removal experiments by Boaventura *et al*., (2003), indicate that grazing, in combination with physiological tolerances, limits the upper shore extent of biotopes characterized by red algal turfs on moderately and more exposed shores, where grazing is greater than on sheltered shores (Hawkins & Hartnoll, 1983, Boaventura *et al*., 2003). These results concord with other studies that show grazing and emersion stress limit the height to which red algal turfs can extend (Underwood, 1980; Boaventura, 2000).

The green algae are considered resistant to this pressure (although may be bleached at higher shore levels during periods of high temperature) as they are found throughout the intertidal including the high shore levels which may not be inundated every day.

Mobile epifauna are likely to relocate to more suitable habitats. Species such as *Patella vulgata* and *Littorina littorea* that are found throughout the intertidal zone are adapted to tolerate desiccation to some extent. For example, littorinids can seal the shell using the operculum while limpets clamped tightly to rock will reduce water loss.

Sensitivity assessment. Other species better able to tolerate desiccation are likely to competitively displace *Fucus serratus* following increased emergence. A significant, long-term, increase in emergence is therefore considered likely to lead to replacement of this biotope with one that is similar but more typical of the changed conditions with less red algae. *Corallina officinalis* and associated red algae are intolerant of desiccation but basal crusts may allow individuals to persist in conditions that are unfavourable to frond development until the emergence regime is re-established. Following a decrease in emergence *Fucus serratus* may be replaced by *Laminaria digitata* leading to biotope reclassification. As emergence is a key factor structuring the distribution of animals on the shore, resistance to a change in emergence (increase or decrease) is assessed as ‘Low’. Recovery is assessed as ‘High’, (based on *Fucus serratus* and red algae and biotope sensitivity is therefore assessed as 'Medium'.

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

This biotope is found in a range of wave exposures including exposed, moderately exposed and sheltered from wave action (Connor *et al*., 2004).

An increase in wave exposure generally leads to a decrease in macroalgae abundance and size (Lewis, 1961; Stephenson & Stephenson, 1972; Hawkins *et al*., 1992; Jonsson *et al*., 2006). Fucoids are highly flexible but not physically robust and an increase in wave exposure can cause mechanical damage, breaking fronds or even dislodging whole algae from the substratum. Fucoids are permanently attached to the substratum and would not be able to re-attach if removed. Organisms living on the fronds and holdfasts will be washed away with the algae whereas free-living community components could find new habitat in surrounding areas. Wave exposure has been shown to limit size of fucoids (Blanchette, 1997) as smaller individuals create less resistance to waves. As exposure increases the fucoid population will become dominated by small juvenile algae, and dwarf forms of macroalgaes which are more resistant to this pressure. An increase in wave action beyond this would lead to a further increase in the abundance of robust fucoids and red seaweeds, such as *Corallina officinalis* (Connor *et al*., 2004).

As water velocity increases red algae 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 wave action levels. 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. On exposed shores, larger, dense patches of *Osmundea pinnatifida* were more able to withstand increased wave action in winter than small patches, which were severely damaged, presumably due to the number and density of stolons (Prathep, 2001).

A decrease in wave exposure may ultimately reduce *Patella vulgata* abundance because the species does not favour thick algal cover that is often present on more sheltered shores. Alternatively, an increase in significant wave height, linked to increased exposure, may result in population changes with fewer macroalgae and with more *Chthamalus* sp. present than *Semibalanus balanoides* (Ballantine, 1961) and the limpet *Patella ulyssiponensis* present, or present in greater numbers, rather than P*atella vulgata* (Thompson, 1980). These changes are not considered to lead to a significant change in biotope character as species replacements are functionally similar.

Sensitivity assessment. The natural wave exposure range of this biotope is considered to exceed changes (increases and decreases) at the pressure benchmark and this biotope is considered to have 'High' resistance and 'High' resilience (by default), to this pressure (at the benchmark). This assessment is supported by evidence for the tolerance and adaptions of the key characterizing macroalgae to different levels of wave exposure.

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.

Little information was found concerning the effects of heavy metals on turf forming and encrusting coralline algae. However, Bryan (1984) suggested that the general order for heavy metal toxicity in seaweeds is: organic Hg> inorganic Hg > Cu > Ag > Zn> Cd>Pb. Most of the information available suggests that the associated adult gastropod molluscs are rather tolerant of heavy-metal toxicity (Bryan, 1984). Littorinids 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) 0.25-5 µg/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 10 µg/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).

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.

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). Intolerance to hydrocarbon pollution has been assessed to be high, as key structural and important characterizing coralline algal species will be lost and the biotope not be recognized in their absence. Hydrocarbon contamination, at levels greater than the benchmark, e.g. from spills of fresh crude oil or petroleum products, may cause significant loss of *Ulva* spp. However, the species tends to recover very rapidly from oil pollution incidents. For instance, after the *Torrey Canyon* tanker oil in 1967, grazing species were killed, and a dense flush of ephemeral green algae (*Ulva*, *Blidingia*) appeared on the rocky shore within a few weeks and persisted for up to one year (Smith, 1968).

In areas of moderate oil deposit, up to about 1/2 cm thick, on rocks after the *Torrey Canyon* oil spill, limpets had survived unscathed over a month after the event and feeding continued even though a coating of oil smothered their food source of algae and diatoms (Smith, 1968). Limpets can ingest thick oil and pass it through their gut. However, thick layers of oil smothering individuals will interfere with respiration and spoil normal food supplies for *Patella vulgata*. 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). Thus *Patella vulgata* has higher intolerance to fresh oil which has a high component of volatile hydrocarbons remaining. A significant reduction in the density of juvenile limpets was also observed at all sites known to have been oiled by the *Sea Empress* spill (Moore, 1997). 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.

Cole *et al*. (1999) suggested that herbicides were (not surprisingly) very toxic to algae and macrophytes. Hoare & Hiscock (1974) noted that with the exception of *Phyllophora* species, all red algae including encrusting coralline forms, were excluded from the vicinity of an acidified halogenated effluent discharge in Amlwch Bay, Anglesey and that intertidal populations of *Corallina officinalis* occurred in significant amounts only 600m east of the effluent. 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.

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

Bay *Patella vulgata* was excluded from sites within 100-150m of the discharge of acidified, halogenated effluent.

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

No evidence.

Introduction of other substances	Not Assessed (NA) Q: NR A: NR C: NR	Not assessed (NA) Q: NR A: NR C: NR	Not assessed (NA) Q: NR A: NR C: NR

This pressure is **Not assessed**.

De-oxygenation	High Q: High A: High C: High	High Q: High A: High C: High	Not sensitive Q: High A: Medium C: Medium

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/or 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.

Reduced oxygen concentrations have been shown to inhibit both photosynthesis and respiration in macroalgae (Kinne, 1977). Despite this, macroalgae are thought to buffer the environmental conditions of low oxygen, thereby acting as a refuge for organisms in oxygen depleted regions especially if the oxygen depletion is short-term (Frieder *et al*., 2012). If levels do drop below 4 mg/l negative effects on these organisms can be expected with adverse effects occurring below 2 mg/l (Cole *et al*., 1999). Reduced oxygen levels are likely to inhibit photosynthesis and respiration but not cause a loss of the macroalgae population directly. However, small invertebrate epifauna may be lost, causing a reduction in species richness.

Experiments have shown that thallus discs of *Ulva lactuca* plants can survive prolonged exposure to anoxia and hypoxia (Vermaat & Sand-Jensen, 1987; Corradi *et al*., 2006). Following resumption of normal oxygen conditions gametes were produced. The associated invertebrate species also 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). *Semibalanus balanoides* can respire anaerobically, so they can tolerate some reduction in oxygen concentration (Newell, 1979). When placed in wet nitrogen, where oxygen stress is maximal and desiccation stress is low, *Semibalanus balanoides* have a mean survival time of 5 days (Barnes *et al*., 1963). 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* are able to respire in air, mitigating the effects of this pressure during the tidal cycle.

Sensitivity assessment. As the biotope will only be exposed to this pressure when submerged during the tidal cycle and wave action will re-oxygenate waters while respiration will occur in air, biotope resistance was assessed as 'High' and resilience as 'High' (no effect to recover from), resulting in a sensitivity of 'Not sensitive'.

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

This pressure relates to increased levels of nitrogen, phosphorus and silicon in the marine environment compared to background concentrations. The benchmark is set at compliance with WFD criteria for good status, based on nitrogen concentration (UKTAG, 2014).

High levels of enrichment may stimulate algal blooms and macroalgal growth. Stimulation of fast-growing green algae may result in smothering of boulder surfaces and lead to negative effects on both perennial algae and bases and the characterizing invertebrate hinder perennial species development or harm their recruitment. Krauflin *et al*. (2006) found only minor effect on the fucoid community structure as a response to high nutrient levels during the first 3 years of an enrichment experiment. During the 4th year of exposure however, *Fucus serratus* started to decline and population consequently crashed in the 5th year. The study observed full recovery of algal canopy and animal community in less than 2 year after conditions returned to normal. The results indicate that established rocky shore communities of perennial algae with associated fauna are able to persist for several years, even at very high nutrient levels, but that community shifts may suddenly occur if eutrophication continues. They also indicate that rocky shore communities have the ability to return rapidly to natural undisturbed conditions after the termination of nutrient enhancement.

The red alga *Corallina officinalis* and the associated green algae species 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. 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. Grazers in the biotope may also benefit from increased availability of food resources, due to enhanced growth.

Atalah & Crowe (2010) added nutrients to rockpools 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. The experimental treatments do not directly relate to the pressure benchmark but indicate some general trends in sensitivity. Nutrients had no significant effect on the cover of *Corallina officinalis*. The cover of green filamentous algae was significantly increased both by reduced grazing and increased nutrients, although the effect size was synergistically magnified by the combined effect of grazer removal and nutrients. Nutrient enrichment caused an absolute increase in the average cover of green filamentous algae of 19% (±3.9 S.E.) respect to the control treatments while the cover of red turfing algae was not affected by nutrient addition (Atalah & Crowe, 2010)

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 red algae, including *Corallina officinalis* e.g. Bellgrove *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’. Grazers limit the effects of nutrient enrichment; in the absence of *Gibbula cineraria*, *Littorinids* and other grazers, significant changes in the structure of the algal assemblage could emerge following eutrophication. However, where boulders move frequently this would prevent permanent colonization by larger, perennial species and ephemeral algae could be displaced by space-occupying red algae following disturbance and the creation of gaps.

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

It should be noted that biotopes occurring in tide swept or wave exposed areas are less likely to experience the effects of organic enrichment as the organic matter will be rapidly removed. Organic matter may be moved to the sediments beneath the boulders and would enhance food supply to deposit feeding infauna such as *Cirratuls cirratulus*, other polychaetes and oligochaetes. At the pressure benchmark the input of organic carbon would represent a food subsidy leading to enrichment rather than gross pollution effects.

Organic enrichment and nutrient enrichment commonly co-occur, for example sewage deposits or outputs from fish farms may enhance nitrogen and phosphorous and organic matter. Nutrient enrichment alters the selective environment by favouring fast growing, ephemeral species such as *Ulva lactuca* and *Ulva intestinalis* (Berger *et al*., 2004; Kraufvelin, 2007). Rohde *et al.* (2008) found that both free growing filamentous algae and epiphytic microalgae can increase in abundance with nutrient enrichment. This stimulation of annual ephemerals may accentuate the competition for light and space and hinder perennial species development or harm their recruitment (Berger *et al*., 2003; Kraufvelin *et al*., 2007). Bellgrove *et al.* (2010) found that coralline turfs out-competed fucoids at a site associated with organic enrichment caused by an ocean sewage outfall. Worldwide *Corallina officinalis* has been noted to increaseat the expense of canopy forming macroalgae within the vicinity of sewage outfalls and at intermediately polluted sites (Bellgrove *et al*., 2010; Littler & Murray, 1975; May, 1985; Brown *et al*., 1990). As turf forming algae *Corallina* spp. trap large amounts of sediment and are therefore 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.

Cabral-Oliveira *et al*., (2014), found higher abundances of juvenile *Patella* sp. and lower abundances of adults closer to sewage input and suggested the structure of these populations was due to increased competition closer to the sewage outfalls.

Sensitivity assessment. Evidence is limited but it is considered that this biotope has ’High’ resistance to increased organic matter at the pressure benchmark (which represents enrichment rather than gross pollution). Resilience is therefore assessed as ‘High’ (no effect to recover from) and the biotope is considered to be 'Not sensitive'.

Physical Pressures

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

All marine habitats and benthic species are considered to have a resistance of ‘None’ to this pressure and to be unable to recover from a permanent loss of habitat (resilience is 'Very low'). Sensitivity within the direct spatial footprint of this pressure is therefore 'High'. Although no specific evidence is described, confidence in this assessment is 'High' due to the incontrovertible nature of this pressure. Adjacent habitats and species populations may be indirectly affected where meta-population dynamics and trophic networks are disrupted and where the flow of resources e.g. sediments, prey items, loss of nursery habitat etc. is altered.

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

This biotope is characterized by the hard rock substratum, provided by the boulders, to which the characterizing *Fucus serratus* and associated species such as red and green algae, barnacles limpets and anemones can firmly attach. A change to a sedimentary substratum (without boulders) would significantly alter the character of the biotope through the loss of boulders and associated changes to the sedimentary assemblage. A biological assemblage more typical of the changed conditions would develop. A change to an artificial hard substratum could also this biotope as it would not be equivalent to a boulder field on sediments.

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 disturbance (Sousa, 1979), microhabitat provision (McGuinness & Underwood, 1986), 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).

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). *Corallina officinalis* shows optimal settlement on finely rough artificial substrata (0.5 - 1 mm 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). Crustose coralline algae extend further to the undersides of natural, rounded boulders than experimental stone blocks (Liversage, 2016). This pattern is likely due to availability of light (Liversage, 2016).

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 or artificial, stable hard substratum would remove the habitat for this biotope. 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)	Low Q: Low A: NR C: NR	Very Low Q: High A: High C: High	High Q: Low A: Low C: Low

A change in sediment type below the boulders would be very likely to alter the biological assemblage as soft sediment infauna are highly sensitive to changes in sediment type. Changes in the size and shape and other characteristics of boulders would also lead to changes in the habitat. Boulders provide three microhabitats, the upper and lower surface of boulders and the substratum underneath the boulders (Cruz Motta *et al*., 2003).

Sensitivity assessment. A change in boulder and sediment characteristics are likely to lead to changes in species composition, and shifts in abundance and distribution of species that remain. The biotope classification is unlikely to change while the species composition is relatively similar. Biotope resistance is assessed as ‘Low’ and resilience is Very low (the pressure is a permanent change), biotope sensitivity is, therefore, assessed as ‘High’.

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

Removal of boulders would result in loss of this habitat. Biotope resistance is assessed as ‘None’ and resilience as ‘Medium’ following restoration of the habitat. Sensitivity is, therefore, assessed as 'Medium'.

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

The species characterizing this biotope occur on the rock and therefore have no shelter from abrasion at the surface.

Most macroalgae are very flexible but not physically robust. Fucoid algae are particularly intolerant of trampling, depending on intensity. Fucoid algae demonstrate a rapid (days to months) detrimental response to the effects of trampling, depending on species, which has been attributed to either the breakage of their fronds across rock surfaces (Boalch *et al*., 1974) or their possession of small discoid holdfasts that offer little resistance to repeated impacts (Brosnan & Crumrine, 1992; Fletcher & Frid, 1996b). Foliose species such as *Mastocarpus* spp. are also likely to be intolerant of trampling (Brosnan & Crumrine, 1994). Brosnan (1993) suggested that the presence or absence of foliose algae (e.g. fucoids) could be used to indicate the level of trampling on the rocky shores of Oregon.

In the UK, Boalch *et al.* (1974) and Boalch & Jephson (1981) noted a reduction in the cover of fucoids at Wembury, south Devon, when compared to surveys conducted by Colman (1933). The size ranges of *Ascophyllum nodosum*, *Fucus vesiculosus* and *Fucus serratus* were skewed to smaller length, and the abundance of *Ascophyllum nodosum* in particular was reduced (Boalch & Jephson, 1981). It was suggested that visitor pressure, especially after the construction of a car park, was responsible for the reduced cover of fucoids (Boalch *et al*., 1974). They suggested that the raised edges of the slatey rock severed fronds when the rocks were walked over. However, no quantitative data was provided. Conversely, algal turfs seem to be relatively tolerant of the direct effects of trampling (based on the available evidence) and some species may benefit from removal of canopy forming algae (Tyler-Walters, 2005). Their tolerance may result from their growth form as has been shown for vascular plants and corals (Liddle, 1997). Brosnan (1993) suggested that algal turf dominated areas (on shores usually dominated by fucoids) were indicative of trampling on the rocky shores of Oregon. However, tolerance is likely to vary with species and their growth form and little species specific data was found. Furthermore, algal turfs may suffer negative indirect effects where they form an understorey below canopy forming species.

Pinn & Rodgers (2005) compared a heavily visited ledge with a less visited ledge at Kimmeridge Bay, Dorset. Although the mean species richness was similar at both sites, the total number of species was greater at the less utilized site. Comparatively, the heavily utilized ledge displayed a reduction in larger, branching algal species (e.g. *Fucus serratus*) and increased abundances of ephemeral and crustose species (e.g. *Ulva linza* and *Lithothamnia* spp. respectively). Fletcher and Frid (1996a and b) examined the effects of persistent trampling on two sites on the north east coast of England. The trampling treatments used were 0, 20, 80, and 160 steps per m^2 per spring tide for 8 months between March and November. Using multivariate analysis, they noted that changes in the community dominated by fucoids (*Fucus vesiculosus*, *Fucus spiralis* and *Fucus serratus*) could be detected within 1 to 4 months of trampling, depending on intensity. Intensive trampling (160 steps/m^2 /spring tide) resulted in a decrease in species richness at one site. The area of bare substratum also increased within the first two months of trampling but declined afterwards, although bare space was consistently most abundant in plots subject to the greatest trampling (Fletcher & Frid, 1996a, 1996b). The abundance of fucoids was consistently lower in trampled plots than in untrampled plots.

Fletcher & Frid (1996a and b) also reported a decrease in the understorey algal community of encrusting coralline algae and red algae, which was probably an indirect effect due to increased desiccation after removal of the normally protective fucoid canopy (see Hawkins & Harkin, 1985) by trampling. They also noted that opportunistic algae (e.g. *Ulva* sp.) increased in abundance. Fletcher and Frid (1996a) noted that the species composition of the algal community was changed by as little as 20 steps per m^2 per spring tide of continuous trampling since recolonization could not occur. A trampling intensity of 20 steps per m^2 per spring tide could be exceeded by only five visitors taking the same route out and back again across the rocky shore in each spring tide. Both of the sites studied receive hundreds of visitors per year and damage is generally visible as existing pathways, which are sustained by continuous use (Fletcher & Frid, 1996a, 1996b). However, the impact was greatest at the site with the lower original abundance of fucoids.

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%. *Iridaea cornucopiae* decreased from 38 to 14% cover within a month and continued to decline to 4-8% cover. 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 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 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.

Schiel & Taylor (1999) reported the death of encrusting corallines one month after trampling due to removal of their protective canopy of fucoids by trampling (10 -200 tramples where one trample equals one transect walked by one person). A higher proportion of corallines died back in spring treatments presumably due to the higher levels of desiccation stress expected at this time of year. However, encrusting corallines increased within the following year and cover returned to control levels within 21 months (Schiel & Taylor, 1999). Mechanical abrasion from scuba divers was also reported to impact encrusting corallines, with cover of *Lithophyllum stictaeforme* greater in areas where diving was forbidden than visited areas (abundance, 6.36 vs 1.4; it is presumed this refers to proportion of cover, although this is not clear from the text) (Guarinieri *et al*., 2012).

Dethier (1994) experimentally manipulated surface abrasion on a range of encrusting algae including *Lithophyllum impressum*. Crusts were brushed with either a nylon or steel brush for 1 minute a month for 24 months. Unbrushed controls grew by approximately 50% where the cover of nylon brushed crusts and steel brushed crusts decreased by approximately 25% and 40% respectively (interpreted from figures in Dethier, 1994). In laboratory tests on chips of *Lithophyllum impressum* brushing with a steel brush for 1 minute once a week for 3 weeks, resulted in no cover loss of two samples while a third 'thinned and declined' (Dethier, 1994).

Ulva spp. fronds are very thin and could be torn and damaged and individuals may be removed from the substratum, altering the biotope through changes in abundance and biomass. *Ulva* spp. cannot repair damage or reattach but torn fronds could still photosynthesise and produce gametes. Tearing and cutting of the fronds has been shown to stimulate gamete production and damaged plants would still be able to grow and reproduce.

Soft bodied species, such as the anemone *Actinia equina*, are likely to be damaged or removed by abrasion, although anemones may repair damage and fragments may regrow. 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/hrs) 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 3 mm long) were crushed. However, on the same shore, the authors found that limpets may be relatively more resistant to abrasion from trampling. Following step and kicking experiments, few individuals of the limpet 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 available evidence indicates that abrasion could cause a significant loss of fucoid cover and a reduction in species abundance and diversity. Based on the available evidence it is concluded that the biotope is sensitive to abrasion and that resistance of characterizing and associated species (excluding *Corallina officinalis*) is 'Low' (loss of 25-75% of bed within direct impact footprint), resilience is assessed as 'Medium', resulting in a sensitivity of 'Medium'.

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

Disturbance that leads to movement and overturning of boulders is a key factor structuring the community. Disturbance can affect species numbers under boulders (McGuiness, 1987). Disturbance can also affect the pattern of algal cover on boulders (Addessi, 1994) and overturning is a key factor in determining species richness (Sousa, 1979). Boulders that are subject to high levels of disturbance and over-turning are colonized by fast growing green algae and disturbance resistance species while large boulders that are rarely over-turned host late successional communities (Sousa, 1979). The biotope LR.MLR.BF.Fser.Bo as described (Connor *et al*., 2004), is species rich and is unlikely to be subject to frequent disturbance. A single event of overturning, where the upper surface is turned on to the sediment is likely to lead to loss of algae as these will be smothered and damage and mortality of attached epifauna. Some under-boulder species such as amphipods and polychaetes may be able to relocate back to the underside of the boulder.

Sensitivity assessment. A single event of disturbance leading to the boulder being overturned is likely to alter species composition. Biotope resistance is assessed as ‘Low' and resilience as ‘Medium’, biotope sensitivity is therefore ‘Medium’.

Changes in suspended solids (water clarity)	Low Q: High A: Medium C: Medium	High Q: High A: High C: High	Low Q: High A: Medium C: Medium

Intertidal biotopes will only be exposed to this pressure when submerged during the tidal cycle and thus have limited exposure. Siltation, which may be associated with increased suspended solids and the subsequent deposition of these is assessed separately (see siltation pressures). In general, increased suspended particles reduce light penetration and increase scour and deposition. Changes in suspended solids 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 deposit feeding infauna, such as *Cirratulus cirratulus*, that are present in the sediment below boulders and in trapped sediments amongst the algae are not likely to be directly sensitive to increased or decreased turbidity.

Changes in suspended solids affecting water clarity will have a direct impact on photosynthesis in *Fucus serratus*. Irradiance below the light compensation point of photosynthetic species can compromise growth (Middelboe *et al*., 2006). However, turbidity is only relevant when the biotope is covered with water as seaweed photosynthesis declines on emersion and recommences when recovered with water. Increased siltation may cover the frond surface of *Fucus serratus* and other macroalgae with a layer of sediment further reducing photosynthesis and growth rate. Sediment deposition can also interfere with attachment of microscopic stages of seaweeds reducing recruitment (see ‘siltation’ pressures). In extreme turbidity, such as found in the Bristol Channel, *Fucus serratus* is excluded from the bottom of the intertidal (below 2 m above chart datum) due to the lack of light for sustained growth (Chapman, 1995).

Increases in the cover of sediment trapping, turf forming red 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). As turfs of *Osmundea pinnatifida* trap sediments (Prathep *et al*., 2003), it is clear that this species has some resistance to abrasion and scour from sediment particles. *Corallina* species accumulate more sediment than any other alga (Hicks, 1985). Hence an increase in suspended sediment is likely to accumulate in the patches of *Corallina officinalis*. A significant increase may result in smothering (see siltation pressures). 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 also result in increased scour, which may adversely affect *Fucus vesiculosus* and foliose red algae, and interfere with settling spores and recruitment if the factor is coincident with their major reproductive period. However, 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.

The biotope occurs in shallow waters 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. Limited shading from suspended sediments is therefore not considered to negatively affect this genus. *Palmaria palmata* is often found under partially shaded conditions as an epiphyte on the stems of *Laminaria* spp. (Morgan *et al*. 1980) in the sublittoral zone (Lüning, 1990). In the Bay of Fundy where the tidal flux of nutrients from the marshes includes a high level of suspended sediment, *Palmaria palmata* grows well despite high turbidity. Irvine (1983), observed morphological adaptation of the plant in fairly sheltered, silty conditions; sometimes the blade divisions are wedge-shaped and finely dissected above or the blade has numerous linear divisions throughout. It is likely that this form reduces possible smothering that may result from increased siltation resulting from increased levels of suspended sediments. In the absence of nutrients short-term increase in turbidity may affect growth and reproduction, however, as a perennial, the adults will probably survive. Other red algal species have high tolerances for high levels of suspended solids. *Chondrus crispus* occurs in areas of sand covered rock in the subtidal biotope IR.HIR.KSed.ProtAhn suggesting it is very resistant to high levels of turbidity and scour associated with high levels of resuspended particles.

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.

Experiments have shown that *Ulva* is a shade tolerant genus and can compensate for reduced irradiance by increasing chlorophyll concentration and light absorption at low light levels. *Ulva* spp. were able to survive over two months in darkness and to begin photosynthesising immediately when returned to the light (Vermaat & Sand-Jensen, 1987). Limited shading from suspended sediments is therefore not considered to negatively affect this genus.

Sensitivity assessment. The exposure of this biotope to suspended sediments in the water column will be limited to immersion periods, and wave action will reduce accumulation. The biotope is considered to be 'Not sensitive' to a reduction in suspended solids, although this may reduce food supply to barnacles and other filter and deposit feeders that occur in this biotope. An increase in suspended solids may lead to some sub-lethal abrasion of fronds of *Fucus serratus* and some reduction in photosynthesis while submerged with some effects on recruitment. Evidence globally indicates that increase suspended solids favour the turf-forming algae that occur within this biotope (Airoldi, 2003). Resistance is therefore assessed as 'Low-Medium' and resilience as 'High' so that sensitivity of the biotope is considered to be 'Low'. An increase in suspended solids above the pressure benchmark may result in a change in species composition with an increase in species seen in very turbid, silty environments e.g. *Ahnfeltia plicata*, *Rhodothamniella floridula*, *Polyides rotunda* and *Furcellaria lumbricalis*.

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

Sedimentation can directly affect assemblages inhabiting rocky shores in different ways, but the main direct effects are burial/smothering and scour/abrasion of organisms. The degree of smothering experienced by the characterizing species will be influenced by the size and shape of the boulders. In areas with greater water flow or wave action, excess sediments will be removed from the rock surface within a few tidal cycles and may be redeposited on the sediments below the boulders, although scour patterns around boulders will determine the movement and removal of sediments.

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 could 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), for example, 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.

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 <5 mm 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 ouputs, 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 colonization by sand adapted species (Huff & Jarett, 2007). The community had recovered one month after sand deposition ceased (Huff & Jarett, 2007).

In a review of the effects of sedimentation on rocky coast assemblages, Airoldi (2003) outlined the evidence for the sensitivity of coralline algae to sedimentation. The reported results are contradictory with some authors suggesting that coralline algae are negatively affected by sediments while others report that encrusting corallines are often abundant or even dominant in a variety of sediment impacted habitats (Airoldi, 2003 and references therein). Crustose corallines have been reported to survive under a turf of filamentous algae and sediment for 58 days (the duration of experiment) in the Galapagos (species not identified, Kendrick, 1991). The crustose coralline *Hydrolithon reinboldii* has also been reported to survive deposition of silty sediments on subtidal reefs off Hawaii (Littler, 1973).

Atalah & Crowe (2010) added sediment to rockpools in controlled experiments that appear to be very similar to this biotope. 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*. Sediment treatment involved the addition of a mixture of coarse and fine sand of either 300 mg/cm^2/month or 600 mg/cm^2 every 15 days (the depth of sediment was not reported). 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 both levels of sediment led to a significant decrease in grazers and crustose coralline algae also decreased. Sedimentation had no significant effect on the cover of green filamentous algae (*Ulva* sp.) but led to an increase in the mean cover of red turfing algae (*Mastocarpus stellatus* and *Chondrus crispus* and *Corallina officinalis*) from 11.7% (±1.0 S.E.) in controls to 26.1% (±4.7 S.E.) in sedimented assemblages, but there were no differences between the two levels of sedimentation. The cover of red filamentous algae (*Ceramium* spp. and *Gelidium* spp.) was also significantly increased in the sedimentation experiments. The experimental results support the general trend of greater sensitivity of grazers and encrusting corallines to sedimentation than turf-forming algae.

Observations and experiments indicate that *Ulva* spp. have relatively high tolerances for the stresses induced by burial such as darkness, hypoxia and exposure to sulphides (Vermaat & Sand-Jensen, 1987; Corradi *et al*., 2006; Kamermans *et al*., 1998). *Ulva lactuca* is a dominant species on sand-affected rocky shores in New Hampshire (Daly & Mathieson, 1977), although Littler *et al*. (1983) suggest that *Ulva* sp., are present in areas periodically subject to sand deposition not because they are able to withstand burial but because they are able to rapidly colonise sand-scoured areas. *Ulva* spp. have, however, been reported to form turfs that trap sediments (Airoldi, 2003, references therein) suggesting that resistance to chronic rather than acute siltation events may be higher.

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 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 will 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. Mortality will be reduced, and possibly avoided, where the smothering sediment is removed due to wave action or tidal streams, depending on how long the sediment remains. Resistance has been assessed as ‘Medium’ as boulders are likely to project above the sediment surface and deposited sediments may be removed rapidly. Resilience is assessed as 'High'. Overall the biotope has a ‘Low sensitivity to smothering at the level of the benchmark. It should be noted that the associated *Patella vulgata* and littorinids may have higher sensitivities to this pressure.

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

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. The degree to which boulders are smothered will depend on the size and shape. Where boulders are large they may project above the deposit although sediment re-suspension and scour during the tidal cycle will impact the epifauana and flora. A deposit at the pressure benchmark would cover sediments with a thick layer of fine materials and scour patterns around boulders will influence the time for the sediment to be removed.

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 the *Fucus serratus* understorey algae and invertebrates present on the boulders and within the sediments underlying the boulders. 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 is assessed as having ‘Medium’ sensitivity to siltation, at the pressure benchmark.

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

Increased levels of diffuse irradiation correlate with increased growth in macroalgae (Aguilera *et al*., 1999). Macroalgae require light to photosynthesis so that changes in light intensity are likely to affect photosynthesis, growth, competition and survival. Chapman (1995) noted that too little or too much light are likely to be stressors. There is considerable literature on the light compensation point of marine algae (see Lüning, 1990) but it is difficult to correlate such evidence with 'shading', as light saturation and compensation points depend on light availability, light quality, season and turbidity. As fucoids are out-competed in sublittoral conditions, it is likely that permanent shading would affect their growth and allow them to be out-competed by other, more shade tolerant species, within the affected area.

Corallina officinalis and other red algae are shade tolerant, often occurring under a macroalgal canopy that reduces light penetration. In areas of higher light levels, the fronds may be lighter in colour due to bleaching (Colhart & Johansen, 1973). Other red algae in the biotope are flexible with regard to light levels. Canopy removal experiments in a rocky subtidal habitat in Nova Scotia, Canada by Schmidt & Scheibling (2007) did not find a shift in understorey macroalgal turfs (dominated by *Corallina officinalis*, *Chondrus crispus* and *Mastocarpus stellatus*) to more light-adapted species over 18 months. *Coralline* crusts and *Corallina officinalis* are shade tolerant algae, often occurring under a macralgal canopy that reduces light penetration. These species can acclimate to different levels of light intensity and quality and encrusting corallines can occur in deeper water than other algae where light penetration is limited. Samples of *Lithophyllum impressum* suspended from a raft and shaded (50-75% light reduction) continued to grow over two years (Dethier, 1994). In areas of higher light levels, the fronds and bases may be lighter in colour due to bleaching (Colhart & Johansen, 1973). Other red algae in the biotope are flexible with regard to light levels and can also acclimate to different light levels. Canopy removal experiments in a rocky subtidal 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 fucoids are out-competed in sublittoral conditions, it is likely that permanent shading would affect their growth and allow them to be out-competed by other, more shade tolerant species, such as red algae within the affected area. The loss of *Fucus serratus* would lead to biotope reclassification, therefore a biotope resistance of 'Low' is suggested, with low confidence. Resilience is assessed as 'High' so that sensitivity is 'Low'.

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

No direct evidence was found to assess this pressure. The associated macroalgae (with the exception of *Ulva* spp.) have limited dispersal, barriers and changes in tidal excursion are not considered relevant to these species as dispersal is limited.

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

The key characterizing species *Fucus serratus* is not currently cultivated or translocated. 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. No evidence was found for the effects of gene flow between cultivated species and wild populations. *Palmaria palmata* may be cultivated for use as biofilters and/or food. 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. The pressure is considered to be 'Not relevant'.

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

Invasive non-indigenous species (INIS) that can alter habitats (ecological engineers), or out-compete the native species for space and other resources such as light and nutrients, are the most likely species to negatively affect this biotope. Space pre-emption by *Fucus serratus* and the turf and crustose bases of the red macroalgae, as well as the trapped sediment within the turf, may prevent settlement of INIS until disturbance events, such as boulder movement, create gaps for invasion. Not all species require bare patches, however, and 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).

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), rather than the exposed shores characteristic of this biotope. The red seaweeds *Heterosiphonia japonica* and *Neosiphonia harveyi* may also occur in this biotope but again no impacts have been reported. 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.

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. It is found from the mid-littoral to the upper subtidal, and grows on hard substrata but also on other bivalves (e.g. blue mussels) and polychaete reefs (Padilla, 2010). Diederich (2005, 2006) examined settlement, recruitment and growth of *Magallana gigas* (as *Crassostrea gigas*) in the northern Wadden Sea. Recruitment of *Magallana gigas* was significantly higher in the intertidal than the shallow subtidal and the survival of adult oysters in the subtidal was limited by predation. *Magallana gigas* prefer to settle on conspecifics, so that it can build massive oyster reefs.

Dense aggregations of *Magallana gigas* on a former mussel bed showed increased abundance and biomass of *Littorina littorea* in the Wadden Sea (Markert *et al*., 2010). However, Eschweiler & Buschbaum (2011) found that juvenile *Littorina littorea* could carry *Magallana gigas* and *Crepidula fornicata* as epibionts. Body dry weight of snails without oyster overgrowth was twice as high compared to winkles covered with oysters. Also crawling speed of snails with oyster epigrowth was significantly slowed down and about ten times lower than in unfouled periwinkles. Additionally, oyster epibionts caused a strong decrease in reproductive output. In laboratory experiments, egg production of fouled *Littorina littorea* was about 100-fold lower than in affected individuals. Field surveys in different years and habitats demonstrated that up to 10% of individuals occurring on epibenthic bivalve beds and up to 25% of snails living on sand flats may be fouled by *Magallana gigas*.

Thompson & Schiel (2012) found that native fucoids show high resistance to invasions by the Japanese kelp *Undaria pinnatifida*. However cover of *Fucus serratus* was inversely correlated with the cover of the invasive *Sargassum muticum* indicating competitive interaction between the two species (Stæhr *et al*., 2000). Stæhr *et al*. (2000) determined that the invasion of *Sargassum muticum* could affect local algal communities through competition mainly for light and space. Hammann *et al.* (2013) found that in the Baltic Sea, *Gracilaria vermiculophylla* could impact *Fucus vesiculosus* through direct competition for recourses, decreasing the half-life of germlings, and increasing the level of grazing pressure. To date *Gracilaria vermiculophylla* has only been recorded in Northern Ireland, and not on mainland Britain. The introduction of this species to intertidal rocky shores around the British Isles could have negative impacts on native fucoids, and could become relevant to this specific biotope.

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 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. If predation of littorinids was significantly increased this could impact the algal composition and abundance of this biotope by altering the level of grazing pressure.

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 represent 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 recognizable from the description. Based on *Crassostrea gigas* biotope 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 recognize that recovery may be prolonged, resilience is assessed as 'Very Low' and sensitivity is therefore assessed as 'High'.

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

Very little is known about infections in *Fucus* (Wahl *et al*., 2011). Coles (1958) identified parasitic nematodes that caused galls on *Fucus serratus* in the south west of Britain. But to date, no mortalities have been associated to the introduction of microbial pathogens. Torchin *et al.* (2002) suggests that there is potential for increased biotic interactions with parasites or pathogens in many marine systems. More recently, Zuccaro *et al*. (2008) detected a number of fungal species associated with *Fucus serratus*. So far no mortalities have been associated with the introduction of microbial pathogens. However, the potential for increased biotic interactions involving parasites or pathogens is on the rise in many marine systems (Torchin *et al*., 2002).

Several coralline and non-coralline species are epiphytic on *Corallina officinalis*. Irvine & Chamberlain (1994) cite tissue destruction caused by *Titanoderma corallinae*. However, no information on pathogenic organisms in the UK was found. *Corallina officinalis*, like many other algal species has been demonstrated to produce antibacterial substances (Taskin *et al*., 2007). No evidence was found for pathogens of the other red algae which may be present in this biotope. The fungal pathogen, *Petersenia palmaria* n. sp (Oomycetes) which infects *Palmaria mollis*, does not affect *Palmaria palmata* (Meer & Pueschel, 1985).

Other species associated with this biotope such as littorinids, patellid limpets and barnacles experience low levels of infestation by pathogens but mass-mortalities have not been recorded. For example, parasitism by trematodes may cause sterility in *Littorina littorea*. *Littorina littorea* are also parasitized by the boring polychaete, *Polydora ciliata* and *Cliona* sp., which weakens the shell and increases crab predation. *Semibalanus balanoides* is considered to be subject to persistent, low levels of infection by pathogens and parasites. Barnacles are parasitized by a variety of organisms and, in particular, the cryptoniscid isopod *Hemioniscus balani*, in which heavy infestation can cause castration of the barnacle. At usual levels of infestation these are not considered to lead to high levels of mortality. Diseased encrusting corallines were first observed in the tropics in the early 1990's when the bacterial pathogen Coralline Lethal Orange Disease (CLOD) was discovered (Littler & Littler, 1995). All species of articulated and crustose species tested to date are easily infected by CLOD and it has been increasing in occurrence at sites where first observed and spreading through the tropics. Another bacterial pathogen causing a similar CLOD disease has been observed with a greater distribution and a black fungal pathogen first discovered in American Samoa has been dispersing (Littler & Littler, 1998). An unknown pathogen has also been reported to lead to white 'target-shaped' marks on corallines, again in the tropic (Littler *et al*., 2007). No evidence was found that these are impacting temperate coralline habitats.

Sensitivity assessment. No evidence was found that outbreaks of microbial pathogens significantly impact populations of the key characterizing *Fucus serratus* and other associated algal species. Limpets, barnacles and littorinids may be subject to persistent low levels of infestation by pathogens but these are not recorded to lead to high-levels of mortality. Biotope resistance is, therefore, assessed as ‘High’ and resilience as ‘High’, so that the biotope is considered to be ‘Not sensitive’.

Removal of target species	Low Q: High A: High C: High	High Q: High A: Medium C: Medium	Low Q: High A: Medium 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. Many of the species characterizing or associated with this biotope may be targeted by either recreational or commercial harvesters.

Following harvesting the loss of the fucoid company has both short and long-term consequences for benthic communities in terms of diversity, resulting in shifts in community composition and a loss of ecosystem functioning such as primary productivity (Lilley & Schiel, 2006; Gollety *et al*., 2008). Removal of the protective fucoid canopy can increase desiccation and the temperatures experienced by the understorey red algae and invertebrates. This can result in changes in distribution as species either move away or die-back.

Red algae within the biotope may also be subject to hand gathering. *Mastocarpus stellatus* is harvested commercially in Scotland and Ireland to produce carrageen, 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. *Corallina officinalis* is collected for medical purposes; the fronds are dried and converted to hydroxyapatite and used as bone forming material (Ewers *et al*., 1987). It is also sold as a powder for use in the cosmetic industry.

Littorinids are one of the most commonly harvested species of the rocky shore. Large scale removal of *Littorina littorea* may allow a proliferation of opportunistic green algae, such as *Ulva*, on which it preferentially feeds. Experiments designed to test the effects of harvesting by removing individuals at Strangford Lough found that there was no effect of experimental treatments (either harvesting or simulated disturbance) on *Littorina littorea* abundance or body size over a 12 week period (Crossthwaite *et al*. 2012). This suggests that these animals are generally abundant and highly mobile; thus, animals that were removed were quickly replaced by dispersal from surrounding, un-harvested areas. However, long-term exploitation, as inferred by background levels of harvest intensity, did significantly influence population abundance and age structure (Crossthwaite *et al*., 2012). A broadscale study of harvesting in Ireland using field studies and interviews with wholesalers and pickers did suggest that some areas were over harvested but the lack of background data and quantitative records make this assertion difficult to test (Cummins *et al*., 2002). Changes in grazer abundance can alter the character of the algal assemblage. Grazer removal (manual removal of all gastropods in pool and a 1 m 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 (25.4%) (Atalah & Crowe, 2010).

Sensitivity assessment As the species that are harvested in this biotope are attached, sedentary or slow moving and relatively conspicuous a single event of targeted harvesting could efficiently remove individuals and resistance is assessed as ‘Low’. Resilience of *Fucus serratus*, the turf forming red seaweeds and littorinids 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.

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

Direct, physical impacts are assessed through the abrasion and penetration of the seabed pressures, while this pressure considers the ecological or biological effects of incidental removal of species. Removal of a large proportion of the brown and red macroalgae would substantially alter the character of the biotope. Loss of the characterizing and associated species would also alter ecosystem functions, such as rates of production and the provision of a structurally complex habitat.

Sensitivity assessment. Removal of species as by-catch would remove the biological assemblage that defines the biotope, hence the biotope is considered to have ‘Low’ resistance to this pressure and to have ‘Medium’ resilience. Biotope sensitivity is therefore ‘Medium’.

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