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Seaweeds in sediment-floored eulittoral rockpools

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

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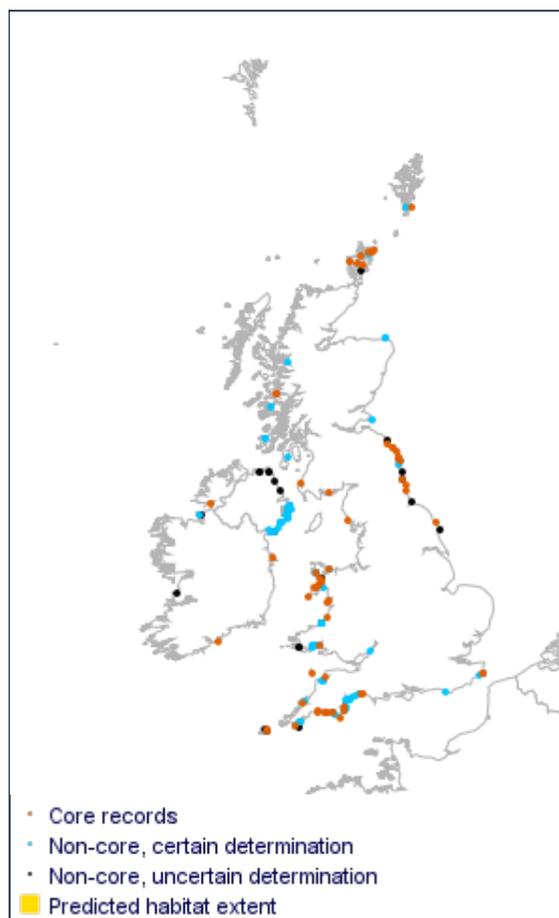
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Seaweeds in sediment (sand or gravel)-floored eulittoral rockpools.

Photographer: Tom Mercer

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

Researched by Frances Perry Referred by Admin

Summary

☰ UK and Ireland classification

EUNIS 2008	A1.413	Seaweeds in sediment-floored eulittoral rockpools
JNCC 2015	LR.FLR.Rkp.SwSed	Seaweeds in sediment-floored eulittoral rockpools
JNCC 2004	LR.FLR.Rkp.SwSed	Seaweeds in sediment-floored eulittoral rockpools
1997 Biotope	LR.LR.Rkp.SwSed	Seaweeds in sediment (sand or gravel)-floored eulittoral rockpools

🔍 Description

Rockpools with sediment floors support distinct communities of scour-tolerant algae. Deep pools with sediment are similar to LR.FK, and are typically dominated by fucooids and kelps (*Fucus serratus*, *Laminaria digitata*, *Saccharina latissima* and *Saccorhiza polyschides*). Areas of hard substrata near to the interface with the sediment are, however, characterized by a range of sand-tolerant algae such as *Furcellaria lumbricalis*, *Polyides rotunda*, *Ahnfeltia plicata* and *Rhodothamniella floridula*

(compare with LR.FK). *Chorda filum* may occur attached to pebbles and shells embedded within the sediment. In pools with large areas of sand, infaunal species such as *Arenicola marina* and *Lanice conchilega* often occur. The seagrass *Zostera* spp. may occur in some pools where stable sand is present. Shallow rockpools with cobble and pebble floors, often with an underlying layer of sediment, support red algal tufts consisting of coralline crustm *Corallina officinalis*, *Condrus crispus* and *Mastocarpus stellatus* mixed with *Ceramium* spp., *Calliblepharis ciliata* and *Cystoclonium purpurea* and green algae (*Cladophora*spp. and *Ulva intestinalis*) (Information taken from the Marine Biotope Classification for Britain and Ireland, Version 97.06: Connor *et al.*, 1997a, b, 2004).

The long list of characterising species is partly due to low similarity between the available records and care should be taken not to interpret this solely as a very high species richness.

↓ Depth range

Upper shore, Mid shore, Lower shore, 0-5 m

🏛️ Additional information

Factors such as pool depth, surface area, volume, orientation to sunlight, shading, internal topography, sediment content and type, together with wave exposure, shore height, and hence flushing rate, and the presence of absence of freshwater runoff, results in large spatial variation in community structure, even between adjacent pools at the same shore height (Ganning, 1971; Metaxas & Scheibling, 1993). Seasonal fluctuations in the abundance of ephemeral seaweeds will occur (Connor *et al.*, 2004), and individual rockpools and the communities that occupy them are highly variable. The above biotope description includes a wide variety of rockpool communities, from deep macroalgal dominated pools to shallow sediment filled rockpools that support only a few red algae. As a necessity, therefore, the review that follows is broad in nature.

✓ Listed By

- none -

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Habitat review

🔄 Ecology

Ecological and functional relationships

The rockpool environment varies depending on factors such as pool depth, surface area, volume, orientation to sunlight, shading, internal topography, sediment content and type, together with wave exposure, shore height, and hence flushing rate, and the presence or absence of freshwater runoff. As a result, no two rockpools exhibit exactly the same physio-chemical conditions and exhibit large spatial variation in community structure, even between adjacent pools at the same shore height (Ganning, 1971; Metaxas & Scheibling, 1993; Metaxas *et al.*, 1994). Any given rockpool is effectively unique (Metaxas & Scheibling, 1993). Therefore, while the organisms detailed below have been recorded in this rockpool biotope, not all, or in some case few, of the organisms, and hence the relationships between them, may occur in any one rockpool.

As communities in rockpools remain constantly submerged and the danger of desiccation is absent, it might be expected that rockpools provide an easier environment in which to live for marine life than drying rock surfaces, and that species from regions lower on the shore would be able to extend much further up the shore. However, the rockpool environment differs from that of the surrounding emergent rock surfaces, so that not all species that thrive on the surrounding rock occur abundantly in rockpools and much of the lower shore open rock fauna is absent from rockpools (Lewis, 1964). Rockpools constitute a distinct environment for which physiological adaptations by the flora and fauna may be required (Lewis, 1964; Metaxas & Scheibling, 1993). The following description is based on reviews by Lewis (1964), Ganning (1971) and Metaxas & Scheibling (1993), the species listed in the MNCR database (JNCC, 1999) and additional references as cited.

- Macroalgae such as kelps, fucoids, red and green algae, erect and encrusting corallines provide primary productivity either directly to grazing invertebrates and fish or indirectly, to detritivores and decomposers, in the form of detritus and drift algae or as dissolved organic material and other exudates. Benthic microalgae and phytoplankton (e.g. diatoms) also add to primary productivity.
- Where present, large macroalgae such as *Halidrys siliquosa* and laminarians (e.g. *Laminaria digitata*, *Saccharina latissima* and *Saccorhiza polyschides*) and fucoids (e.g. *Fucus serratus*, *Fucus vesiculosus*) shade the substratum (depending on density) so that understorey plants tend to be shade tolerant red algae. Understorey algae, by effectively restricting access to the substratum, may also inhibit or restrict recruitment of other species of macroalgae (Hawkins & Harkin, 1985; Hawkins *et al.*, 1992).
- Macroalgae compete for space with sessile invertebrates such as sponges, hydroids, ascidians and bryozoans.
- Macroalgae provide substrata and refuges for a variety of invertebrates and epiphytic algae. The stipes and lamina of *Laminaria* spp. may support bryozoans (e.g. *Membranipora membranacea* or *Electra pilosa*) and grazing blue-rayed limpets (*Helcion pellucida*), while their holdfasts provide additional refuges for meiofauna and small invertebrates. If present, the stipes of *Laminaria hyperborea* may support numerous epiphytes such as *Palmaria palmata*, *Phycodrys rubens* and *Cladophora rupestris* (Goss-Custard *et al.*, 1979). Where present, *Halidrys siliquosa* provide substratum for epiphytes, depending on location, including microflora (e.g. bacteria, blue green algae, diatoms and juvenile larger algae), *Ulothrix* and *Ceramium* sp., hydroids (e.g. *Obelia* spp.), bryozoans (e.g. *Scrupocellaria*

- spp.), and ascidians (e.g. *Apilidium* spp., *Botryllus schlosseri*, and *Botrylloides leachi*) (Lewis, 1964; Moss, 1982; Connor *et al.*, 1997a).
- The macroalgae provide refuges for small invertebrates, such as isopods, amphipods, ostracods and copepods. *Corallina officinalis* provides a substratum for small spirorbids e.g. *Spirorbis corallinae*, which is only found on *Corallina officinalis*. Increasing density of *Spirorbis corallinae* was shown to increase the species richness of the epiphytic fauna (Crisp & Mwiseje, 1989). The invertebrate fauna of *Corallina officinalis* is detailed in ELR.Coff.
 - Amphipods, isopods (e.g. *Idotea granulosa*) and other mesoherbivores graze the epiphytic flora and senescent macroalgal tissue, which may benefit the macroalgal host, and may facilitate dispersal of the propagules of some macroalgal species (Brawley, 1992b; Williams & Seed, 1992). Mesoherbivores also graze the macroalgae but do not normally adversely affect the canopy (Brawley, 1992b).
 - Grazers of periphyton (bacteria, blue-green algae and diatoms) or epiphytic algae include harpacticoid copepods, the limpets *Patella vulgata* and *Patella ulyssiponensis*, the blue-rayed limpet *Patella pellucida*, and gastropods such as *Steromphala cineraria*, *Steromphala umbilicalis*, *Littorina saxatilis*, *Littorina littorea*, *Littorina obtusata* and *Rissoa* spp. Limpets and littorinids also graze macroalgal sporelings and green algae especially.
 - Coralline algae are probably relatively grazing resistant (Littler & Kauker, 1984) and few species graze the corallines directly except perhaps chitons and limpets of the genus *Tectura*.
 - Grazing by littorinids and gammarid amphipods has been shown to significantly affect macroalgal abundance and diversity. For example in cage experiments in littoral fringe pools, Parker *et al.* (1993) found that gammarid amphipods significantly reduced the erect macroalgal canopy, while littorinids grazed microalgae and macroalgal sporelings, and prevented the establishment of erect and encrusting algal canopies. Both groups reduced the species richness of the algal canopy (Parker *et al.*, 1993). In tidepools in Nova Scotia, Chapman (1990) and Chapman & Johnson (1990) reported that grazers (especially littorinids) reduced the abundance of *Fucus* spp. sporelings and juveniles but increased the abundance of ephemeral algae, while having no effect on the encrusting red alga *Hildenbrandia rubra* (Metaxas & Scheibling, 1993). Conversely, Lubchenco (1978) noted that the addition of littorinids to mid-shore pools in Massachusetts decreased the abundance of dominant *Ulva* spp. (as *Enteromorpha* spp.) in favour of *Chondrus crispus* (Metaxas & Scheibling, 1993) a less palatable red alga. Wolfe & Harlin (1988a) noted that Rhode Island tidepools with the highest littorinid densities had the lowest abundance of macroalgae. Similarly, removal of the limpet *Patella vulgata* from high tidal pools at Lough Ine resulted in an increased abundance of *Ulva* (as *Enteromorpha*) sp. (Goss-Custard *et al.*, 1979).
 - Where present, suspension feeders include barnacles (e.g. *Semibalanus balanoides*), the mussel *Mytilus edulis*, hydroids, tubeworms (e.g. *Spirorbis* spp. and *Spirobranchus* spp.), ascidians, bryozoans and sponges. However, the abundance of barnacles and mussels in rockpools is usually low (Lewis, 1964), presumably due to heavy predation on juveniles by the dog whelk *Nucella lapillus* and crabs (e.g. *Carcinus maenas* and *Cancer pagurus*).
 - The sediment provides habitat for deposit feeding annelids e.g. *Arenicola marina* and terebellids.
 - Scavengers include shrimp (e.g. *Palaemon serratus*), brittlestars (e.g. *Amphipholis squamata*), cushion stars (*Asterina gibbosa*) and hermit crabs (e.g. *Pagurus bernhardus*).
 - Invertebrate predators include turbellarians and nemertean feeding on small invertebrates such as copepods and small gastropods. Lower to mid shore rockpools provide refugia for dog whelks *Nucella lapillus* feeding on barnacles and small mussel

within the rockpool and /or leaving the rockpool to forage at high tide. Similarly, crabs such as *Carcinus maenas* and *Cancer pagurus* are generalist predators of gastropods (e.g. littorinids) and bivalves as well as scavengers. Passive carnivores include sea anemones such as *Anemonia viridis*, *Actinia equina* and *Urticina felina*. In addition, intertidal fish such as the shanny *Lipophrys pholis* and gobies prey on small invertebrates such as copepods, amphipods and isopods.

- As with grazing, predation pressure is potentially higher in mid to low shore rockpools, since predators can continue feeding irrespective of the state of the tide. For example, in New South Wales, whelks were shown to reduce the abundance of barnacles, tubeworms and limpets (Fairweather, 1987; Metaxas & Scheibling, 1993). Dethier (1984) concluded that harpacticoid copepod abundance in mid to low shore pools was low due to the presence of predators such as fish and to a lesser extent anemones. The reduced abundance of barnacles and mussels observed in rockpools (Lewis, 1964) is probably partly due to increased predation pressure.

Seasonal and longer term change

Conditions within rockpools are the consequence of prolonged separation from the main body of the sea, and physico-chemical parameters within them fluctuate dramatically (Pyefinch, 1943; Ganning, 1971; Daniel & Boyden, 1975; Goss-Custard *et al.*, 1979; Morris & Taylor, 1983; Huggett & Griffiths, 1986; Metaxas & Scheibling, 1993; Metaxas *et al.*, 1994). In general, larger and deep rockpools low on the shore tend to correspond to the sublittoral habitat with a more stable temperature and salinity regime. In contrast, small and shallow pools higher on the shore are especially influenced by insolation, air temperature and rainfall, the effects of which become more significant towards the high shore, where pools may be isolated from the sea for a number of days or weeks (Lewis, 1964).

Weather conditions exert a considerable influence on temperature and salinity. Water temperature in pools follows the temperature of the air more closely than that of the sea.

Temperature

The temperature of pools fluctuates with air temperature and sunlight, and tend to warm throughout the day, especially if in direct sunlight (Daniel & Boyden, 1975; Goss-Custard *et al.*, 1979). Shaded pools exhibit less dramatic changes in temperature (Daniel & Boyden, 1975). For example, the temperature of an high shore pool exposed to direct sunlight rose quickly in the morning to a maximum of 25 °C, while a shaded high shore pool only rose by 1 °C, even though air temperatures reached 20 °C (Daniel & Boyden, 1975). In addition, deeper pools may become stratified, with warmer water near the surface and cooler near the bottom (Daniel & Boyden, 1975), primarily due to sunlight. They noted that pool temperatures remained almost constant at night and suggested that pool temperatures would fluctuate slowly during the day under overcast conditions. In deeper pools, the vertical temperature gradation present in summer, may reverse during winter owing to density stratification, so that ice may form (Naylor & Slinn, 1958). Morris & Taylor (1983) reported warmer water at the bottom of the pools and cooler at the surface, which they attributed to cooling of the surface water by wind. Examples of temperature ranges reported for mid to low shore pools include annual maxima and minima of 1-25 °C and 2-22 °C (Morris & Taylor, 1983), a diurnal range of 24 °C (day) and 13 °C (night) for a mid shore pool (Daniel & Boyden, 1975), and surface water temperature ranges of 14-19.25 °C and 15.5-20.75 °C in mid shore pools (Pyefinch, 1943). Smaller (or shallow pools) are more affected by climatic change in temperature than larger and/or deeper ones (Ganning, 1971). Morris & Taylor (1983) noted that temperature showed the greatest seasonal variation of all the physical parameters examined. In

summer, the minimum recorded temperatures were greater than the maximum temperatures recorded in winter, and the daily temperature ranges were greater in summer than in winter, in both high and low shore pools (Morris & Taylor, 1983).

Salinity

High air temperatures cause surface evaporation of water from pools, so that salinity steadily increases, especially in pools not flooded by the tide for several days. Alternatively, high rainfall will reduce pool salinity or create a surface layer of brackish/nearly freshwater for a period. In addition, freezing of surface water increases the salinity of the underlying water (Ganning, 1971; Metaxas & Scheibling, 1993). Daniel & Boyden (1975) and Morris & Taylor (1983) reported little variability in salinity over one tidal cycle, and Ganning (1971) suggested that changes in salinity were of limited importance. However, heavy rain resulted in a layer of low salinity water on the surface of pools. Morris & Taylor (1983) stated that a low salinity layer of 2-10 mm was normal but after one storm the low salinity layer increased in depth, eventually resulting in a homogeneous pool of brackish water. Morris & Taylor (1983) reported an annual salinity range in mid to low shore pools of 26-36.5 ppt.

The extent of temperature and salinity change is affected by the frequency and time of day at which tidal inundation occurs. If high tide occurs in early morning and evening the diurnal temperature follows that of the air, whilst high water at midday suddenly returns the temperature to that of the sea (Pyefinch, 1943). Rockpools in the supralittoral, littoral fringe and upper eulittoral are liable to gradually changing salinities followed by days of fully marine or fluctuating salinity at times of spring tide (Lewis, 1964).

Oxygenation

The biological community directly affects oxygen concentration, carbon dioxide concentration and pH, and are themselves affected by changes in the chemical parameters. Throughout the day, algae photosynthesize and produce oxygen, the concentration of which may rise to three times its saturation value, so that bubbles are released (Pyefinch, 1943; Ganning, 1971; Daniel & Boyden, 1975; Goss-Custard *et al.*, 1979; Morris & Taylor, 1983; Metaxas & Scheibling, 1993). In summer, rockpools are likely to be supersaturated with oxygen during the day (Pyefinch, 1943). For example, the greatest range of oxygen saturation of 101.7% occurred in a seaweed dominated, sediment floored pool, which reached over 190% saturation on some days (Pyefinch, 1943). Daniel & Boyden (1975) noted that a mid shore, seaweed dominated pool reached 194% saturation (ca 15 mg O₂/l) but that oxygenation was also marked in shaded pools. A pool with dense fauna exhibited a maximum saturation of 210% (Pyefinch, 1943). During photosynthesis algae absorb carbon dioxide and as concentrations fall, the pH rises. Morris & Taylor (1983) recorded pH values >9 in rockpools on the Isle of Cumbrae.

At night changes occur in the opposite direction as respiration utilizes much of the available oxygen and pH decreases. Daniel & Boyden (1975) reported oxygen depletion at night, with mid to low shore pools reduced to 8-44% saturation. They noted that the crab *Carcinus maenas* leaves the pools at night, and that other species with the ability to air-breathe could also do so, e.g. limpets, littorinids, and the shanny *Lipophrys pholis*. They also observed that shrimps gathered at the edge of high shore pools at night, presumably to take advantage of the better oxygenated surface layer (Daniel & Boyden, 1975). Daily fluctuation in oxygen concentration and pH also vary seasonally, and with the height of the pool on the shore or pool depth (Daniel & Boyden, 1975; Morris & Taylor, 1983; Metaxas & Scheibling, 1993). Morris & Taylor (1983) noted variation in oxygen concentration with pools with pool depth and proximity to pool algae. Again, inundation by the rising tide causes sudden changes in oxygenation, either sudden drops during the day or increases

at night.

Ganning (1971) noted that the temperature of sediment at the bottom of pools showed little variation in temperature, and noted that diurnal fluctuations of greater than 1 °C were rarely observed in bottom sediments of pools deeper than 0.5 m, although a temperature of fall 0.5-1 °C was observed at the water/sediment interface. The presence of sediment in pools increases the risk of scour, which is likely to vary seasonally, increasing in winter storms. In addition, anoxic conditions within the sediment could potentially cause significant and sudden decreases in oxygen levels if the sediment was stirred up by wave action, although no evidence of this effect was found.

Seasonal change in communities

Tidepool community structure has been shown to vary markedly over time, particularly with season (Metaxas & Scheibling, 1993). However, most studies have examined tidepools overseas, or different tidepools communities to those that occur in this biotope (Dethier, 1984; Wolf & Harlin, 1988a,b; Kooistra *et al.*, 1989; Metaxas *et al.*, 1994) so that the species concerned are very different. Seasonal changes in macroalgal cover and diversity were associated with changes in water temperature, light intensity and day length, and reduced grazing pressure from littorinids in winter, although the community types were relatively stable over time in Rhode Island pools (Wolfe & Harlin, 1988a,b). Metaxas *et al.* (1994) noted that sheet forming algae (e.g. *Ulva lactuca*) were found throughout the year, filamentous forms (e.g. *Cladophora* and *Ceramium*) were present in late spring and summer (although other studies found them to be present from late spring to late autumn) while thick leathery and encrusting forms did not vary seasonally. However, macroalgal diversity was lowest in summer and autumn, especially in mid shore pools, probably due to grazing. Metaxas *et al.* (1994) also noted that the abundance of mussels, littorinids and whelks in Rhode Island peaked in summer due to recruitment but varied significantly between pools.

Red algae exhibit seasonal variation in growth and reproduction and red algal turf declines in abundance during the winter months, partly due to die back and abrasion during winter storms. Although protected from wave action in deep pools, macroalgae will be particularly susceptible to damage and abrasion by wave action and winter storms in shallow sediment filled pools. For example, maximum growth of *Furcellaria lumbricalis* occurs in March/April (Austin, 1960b) and release of carpospores and tetraspores occurs in December/January (Bird *et al.*, 1991). Reproductive bodies are present on the gametophytes of *Ahnfeltia plicata* between July and January and mature carposporophytes occur between October and July (Maggs & Pueschel, 1989). However, in the Bristol Channel, Bamber & Irving (1993) noted that the biomass of *Corallina officinalis* increased steadily through spring and summer and began to decline after July. *Mastocarpus stellatus* (as *Gigartina stellata*) was reported have a perennial holdfast, losing many erect fronds in winter, which grow back in spring (Dixon & Irvine, 1977). *Osmundea pinnatifida* also shows seasonal variation in growth, expanding its perennial holdfast in June to September, and producing erect fronds from October onwards reaching a maximum in February to May (Maggs & Hommersand, 1993). *Corallina officinalis* may be overgrown by epiphytes, especially during summer. This overgrowth regularly leads to high mortality of fronds due to light reduction (Wiedemann, pers. comm.). The ephemeral green seaweeds *Ulva intestinalis* and *Ulva lactuca* are likely to be more abundant in summer depending on grazing pressure. In summer, erect and encrusting corallines may be bleached (especially in shallow pools) and lose their pink pigment but in some species, e.g. *Phymatolithon*, this does not necessarily result in death of the plant and pigment may be re-synthesized (Little & Kitching, 1996).

Habitat structure and complexity

As already noted, no two rockpools exhibit exactly the same physio-chemical conditions and exhibit large spatial variation in community structure, even between adjacent pools at the same shore height (Ganning, 1971; Metaxas & Scheibling, 1993; Metaxas *et al.*, 1994). Habitat complexity will vary with macroalgal cover, the dominant macroalgal species, shore height, pool depth and topography. Deeper pools are likely to be more complex than shallow pools. In this biotope, the influence of sediment and scour on the community is likely to increase with the sediment depth (thickness) and/or with the shallowness of the pool, i.e. shallow pools will exhibit simple communities dominated by sediment resistant red algae. However, a variety of intermediate communities could also be represented by this biotope.

In deep pools the underlying rock is likely to be covered by encrusting corallines. Large macroalgal species (e.g. kelps and fucoids) may dominate the surface of the pool. Their depth within the pool (vertical zonation) is limited by self-shading so that only corallines and red algae occur beneath them. The interface between the bottom sediment and the rock surface is likely to support only sand resistant red algae and fauna, e.g. sand-tolerant algae such as *Furcellaria lumbricalis*, *Polyides rotunda*, *Ahnfeltia plicata*, *Rhodothamniella floridula*, and the anemone *Urticina felina*. The sediment may support infauna such as lugworm (e.g. *Arenicola marina*), the sand mason worm *Lanice conchilega*, terebellids and meiofauna. The upper limit of some species of algae within the pool may be limited by the summer surface water temperatures, and or desiccation after evaporation (e.g. corallines). Grazing intensity due to littorinids may also affect the abundance of fleshy macroalgae, so that the pools may be dominated by less palatable red algae (e.g. *Chondrus crispus* and *Mastocarpus stellatus*). Vertical surfaces within deep pools, and crevices or overhangs present, are likely to be dominated by encrusting fauna e.g. the sponges *Halichondria panicea* and *Hymeniacidon perleve*, tubeworms and anemones. The surface of larger stones and pebbles may support tubeworms and the holdfasts of kelps or *Chorda filum*. The holdfasts of kelps and fucoids, and fronds of filamentous species and erect corallines provide refuges for small invertebrates (e.g. amphipods, isopods and small gastropods) or meiofauna (e.g. copepods) (see ecological relationships above). In addition, the shade of macroalgae provide refuges for shrimps (e.g. *Palaemon* spp.) and intertidal fish (e.g. blennies and gobies), while crevices and underboulder habitats provide additional refuges for crabs.

Rockpool species also display zonation patterns, similar to the emergent species. For example, brown algae and corallines are usually dominant in mid to low shore pools, while green algae tend to dominant high shore pools (Metaxas & Scheibling, 1993). Kooistra *et al.* (1989) noted vertical zonation within pools and found that macroalgal communities could be allocated to different depths within pools in the lower or higher parts of the shore. However, the communities studied in Brittany differed markedly from those found in this biotope. Similarly, zonation patterns have also been reported in flatworms, rotifers, oligochaetes, cladocans, copepods, ostracods, barnacles, amphipods, isopods, chironomid larvae and fish (see Metaxas & Scheibling, 1993). *Littorina littorea*, mussels, whelks, limpets and sea urchins tend to dominate in lower shore pools, while other littorinids dominate higher on the shore (Metaxas & Scheibling, 1993). Nevertheless, Metaxas *et al.* (1994) noted that horizontal spatial variability between pools within the same shore height appeared to be as great as variability along the intertidal gradient, and suggested that the physical setting of the pool may be of primary importance in determining the macroalgal abundance. Dethier (1984) examined the effect of natural disturbance rockpool communities in the coast of Washington State. She noted that disturbance such as heat stress in summer and wave action in winter occurred regularly (ca 1.6 times per year per pool). The observed disturbances affected dominant species, so that no one dominant species could occupy all the pools within the tidal range

at any one time. None of the tidal pool assemblages observed were stable over many generations and disturbances resulted in a mosaic of species assemblages within pools in any one region (Dethier, 1984).

Productivity

Although little information on rockpool productivity was found, rocky shore communities are highly productive and are an important source of food and nutrients for members of neighbouring terrestrial and marine ecosystems (Hill *et al.*, 1998; Raffaelli & Hawkins, 1999). Macroalgae exude considerable amounts of dissolved organic carbon which are absorbed readily by bacteria and may even be taken-up directly by some larger invertebrates. Only about 10% of the primary production is directly cropped by herbivores (Raffaelli & Hawkins, 1999). Dissolved organic carbon, algal fragments and microbial film organisms are continually removed by the sea. This may enter the food chain of local, subtidal ecosystems, or be exported further offshore. Rocky shores make a contribution to the food of many marine species through the production of planktonic larvae and propagules which contribute to pelagic food chains. Accumulations of algal debris are also likely in rockpools and such detrital material contributes to overall productivity.

Information specific to the community was not found but Workman (1983) gave an estimate of primary production by microalgal films on the lower shore in the British Isles to be in the region of 100 g C/m²/yr, much of which will be utilized directly by grazers, while primary productivity for fucoids on sheltered shores was estimated to be 1250 g C/m²/yr (Hawkins *et al.*, 1992) and for encrusting corallines to be 1000 g C/m²/yr (Dawes *et al.*, 1991; Raffaelli & Hawkins, 1999). Ganning & Wulff (1970) reported primary productivity values in terms of gross photosynthesis of between 2 and 3.5 O₂/m³/hr in brackish water rock pools dominated by green algae. Overall, deep rockpools with abundant macroalgae are likely to be highly productive mesocosms on the shore. However, shallower pools, with only sparse macroalgal cover due to sediment scour are likely to be far less productive.

Recruitment processes

As with the emergent rock surfaces, variation and unpredictability in recruitment of species contributes to the variation in dominant species within rockpool communities and hence spatial variation between rockpools (Dethier, 1984). Recruitment processes of characteristic or dominant species are described here.

Flora

- Vadas *et al.* (1992) reviewed recruitment and mortality of early post settlement stages of benthic algae. They concluded that grazing, canopy and turf effects were the most important but that desiccation and water movement may be as important for the early stages. The review indicated that recruitment is highly variable and episodic and that mortality of algae at this period is high. Chance events during the early post settlement stages are therefore likely to play a large part in survival.
- *Laminaria hyperborea* and *Laminaria digitata* produce vast numbers of spores, however they need to settle and form gametophytes within about 1 mm of each other to ensure fertilization and therefore may suffer from dilution effects over distance. However, Chapman (1981) demonstrated that substantial recruitment of *Laminaria digitata* plants to areas barren of kelp plants was possible up to 600 m away from reproductive plants. *Saccharina latissima* (studied as *Laminaria saccharina*), is short-lived (2-4 years), reaches

maturity quickly and colonizes bare substratum rapidly (ca 6 months) (Kain, 1975; Kain, 1979; Birkett *et al.*, 1998; see *MarLIN* review) and may be regarded as relatively opportunistic.

- Settlement of algal spores is partly dependant on their motility (if any) and adhesive properties together with preferences for topography (surface roughness), the chemical nature of the substratum and water movement (Norton 1992; Fletcher & Callow 1992). As with all red algae, the spores of *Ahnfeltia plicata*, *Chondrus crispus*, *Furcellaria lumbricalis* and *Polyides rotunda* are non-flagellate and therefore dispersal is a wholly passive process (Fletcher & Callow, 1992). In general, due to the difficulties of re-entering the benthic boundary layer, it is likely that successful colonization is achieved under conditions of limited dispersal and/or minimum water current activity. Norton (1992) reported that although spores may travel long distances (e.g. *Ulva* sp. 35 km, *Phycodrys rubens* 5 km), the reach of the furthest propagule does not equal useful dispersal range, and most successful recruitment occurs within 10 m of the parent plants. It is expected, therefore, that recruitment of *Ahnfeltia plicata*, *Chondrus crispus*, *Furcellaria lumbricalis*, *Polyides rotunda* and the majority of other macroalgae in the biotope would occur from local populations on the surrounding emergent rock but that establishment and recovery of isolated populations would be patchy and sporadic. Scrosati *et al.* (1994) commented that viability of spores of *Chondrus crispus* was low (<30%) and suggested that reproduction by spores probably does not contribute much to maintenance of the intertidal population in Nova Scotia, compared to vegetative growth of gametophytes. However, macroalgae are highly fecund and widespread in the coastal zone so that recruitment may be still be rapid, especially in the rapid growing ephemeral species such as *Ulva* spp., which reproduce throughout the year with a peak in summer. Similarly, *Ceramium* species produce reproductive propagules throughout the year, while *Mastocarpus stellatus* produce propagules from February to December, and exhibit distinct reproductive papillae in summer (Dixon & Irvine, 1977; Burrows, 1991; Maggs & Hommersand, 1993).
- *Lithophyllum incrustans*, representing the encrusting corallines in the biotope, it has been calculated that 1 mm x 1mm of reproductive thallus produces 17.5 million bispores per year with average settlement of only 55 sporelings/year (Edyvean & Ford, 1984). Dispersal is likely to be in excess of 5 km and spores will settle and new colonies will arise rapidly on bare substratum, although the growth rate is slow (2-7 mm per annum - see Irvine & Chamberlain 1994). The erect coralline *Corallina officinalis* has isomorphic sexual (gametophyte) and asexual (sporophyte) stages (see *MarLIN* review). Settlement and development of fronds is optimal on rough surfaces but settlement can occur on smooth surfaces (Harlin & Lindbergh 1977; Wiedemann pers comm.). *Corallina officinalis* settled on artificial substrata within 1 week of their placement in the intertidal in New England summer suggesting that recruitment is high (Harlin & Lindbergh 1977).

Fauna

- Gastropods exhibit a variety of reproductive life cycles. The common limpets *Patella vulgata* and *Patella ulysiponensis*, the top shell *Steromphala umbilicalis*, and *Littorina littorea* have pelagic larvae with a high dispersal potential, although recruitment and settlement is probably variable. However, *Littorina obtusata* lays its eggs on the fronds of fucoids from which hatch crawl-away miniature adults. Similarly, the dog whelk *Nucella lapillus* lays egg capsules on hard substrata in damp places on the shore, from which crawl-always emerge. Therefore, their dispersal potential is limited but probably designed to colonize an abundant food source. In addition, most gastropods are relatively mobile, so that a large proportion of recruitment of available niches would involve migration. *Nucella lapillus* is an

exception, as they generally do not move far, averaging 100 mm /tidal cycle, or between 30 cm or 10 m per year when in the vicinity of an abundant food source (see *MarLIN* reviews for details; Fish & Fish, 1996).

- *Arenicola marina* lays its eggs in its burrows, while adjacent males release sperm at the sediment surface, the resultant juveniles developing within the burrows before migrating upshore. Long distance recruitment of lugworm from sedimentary areas is probably sporadic, and reproduction within a rockpool is only likely if both sexes are present within the same pool. However, *Lanice conchilega* produce dispersive planktonic larvae.
- Many species of mobile epifauna have long lived pelagic larvae and/or are highly motile as adults. Gammarid amphipods brood their embryos and offspring but are highly mobile as adults and probably capable of colonizing new habitats from the surrounding area (e.g. see *Hyale prevostii* review for example). Similarly, isopods such as *Idotea* species brood their young. *Idotea* species are mobile and active swimmers and probably capable to recruiting to new habitats from the surrounding area by adult migration. Hicks (1985) noted that epiphytic harpacticoid copepods lack planktonic dispersive larval stages but are active swimmers, which is therefore the primary mechanism for dispersal and colonization of available habitats. Some species of harpacticoids are capable to moving between low and mid-water levels on the shore with the tide, while in other colonization rates decrease with increasing distance from resident population. Overall immigration and *in situ* reproduction were thought to maintain equilibrium populations exposed to local extinction, although there may be local spatial variation in abundance (see Hicks, 1985).
- Emson & Whitfield (1989) noted that a population of *Amphipholis squamata* from a mid shore rock pool (in south Devon) displayed more seasonal reproduction and short lifespans than other populations studied, which they suggested may be due to the more stressful environmental conditions provided by the rockpool. *Amphipholis squamata* is a simultaneous hermaphrodite, that broods its young in genital bursae. Emson & Whitfield (1989) noted that recruitment occurred in mid-summer, grow in summer and autumn, mature (at a 1.6-1.7 mm, ca 8-12 months old) and reproduce in the following spring, release juveniles in summer and then die in autumn and winter, a lifespan of ca 13 -17 months. Individuals brood up to an average of 10 embryos per individual depending on size, larger individuals brooding more embryos. Regardless of the short lifespan and brooding habit, the population in the rockpool was highly abundant and had maintained its abundance between 1975 and 1987 (Emson & Whitfield, 1989). Although, a brooding species it is widespread with a cosmopolitan distribution suggesting that dispersal occurs by migration of adults.
- Epiphytic and sessile fauna, such as sponges, hydroids, bryozoans and ascidians, have pelagic but short lived larvae with relatively short effective dispersal ranges, depending on the local hydrography. However, most epiphytic species are widespread and ubiquitous and would probably recruit rapidly from adjacent or nearby populations.

Time for community to reach maturity

Development of the community in this biotope, from bare or denuded rock, is likely to follow a similar successional pattern to emergent sheltered rocky shore communities. The loss of grazing species results in an initial proliferation of ephemeral green then fucoid algae, which then attracts mobile grazers, and encourages settlement of other grazers. Limpet grazing reduces the abundance of fucoids allowing barnacles to colonize the shore. After the *Torrey Canyon* oil spill in March 1967, recovery rates were dependant on local variation in recruitment and mortality so that sites varied in recovery rates, for example maximum cover of fucoids occurred within 1-3 years, barnacle abundance increased in 1-7 years, limpet number were still reduced after 6-8 years

and species richness was regained in 2 to >10 years (depending on the shore and the degree of disturbance) (Southward & Southward, 1978; Hawkins & Southward, 1992; Raffaelli & Hawkins, 1999). Where only the rockpool is affected, recruitment is potentially good, especially in motile species. However, variability in rockpool communities and their response to disturbance (recovery) has been attributed to variation and seasonality in recruitment (Metaxas & Scheibling, 1993).

Dethier (1984) noted that few rockpool populations, even of dominant species, remained static over time, based on long-term observations over several years. On the coast of Washington State, partial disturbance (a reduction in abundance or cover) resulted in relatively rapid recovery of the community for example; encrusting corallines recovered in over 2 months and erect corallines attained 87% of their original cover in 2 years. The red algae *Rhodomela* sp. exhibited 39% recovery from total loss after 2 years and *Cladophora* sp. exhibited 77% recovery after total loss. Dethier (1984) concluded that disturbance was a factor that resulted in a mosaic of different communities in rockpools within an area and that, at any point in time, separate rockpool communities were probably in different stages of recovery.

Additional information

None

Preferences & Distribution

Habitat preferences

Depth Range	Upper shore, Mid shore, Lower shore, 0-5 m
Water clarity preferences	
Limiting Nutrients	Data deficient
Salinity preferences	Full (30-40 psu), Reduced (18-30 psu), Variable (18-40 psu)
Physiographic preferences	Enclosed coast / Embayment, Open coast
Biological zone preferences	Eulittoral, Lower eulittoral, Lower littoral fringe, Mid eulittoral, Sublittoral fringe, Upper eulittoral, Upper littoral fringe
Substratum/habitat preferences	Bedrock, Cobbles, Gravel / shingle, Mud, Pebbles, Sand, Small boulders
Tidal strength preferences	Moderately Strong 1 to 3 knots (0.5-1.5 m/sec.), Strong 3 to 6 knots (1.5-3 m/sec.), Very Weak (negligible), Weak < 1 knot (<0.5 m/sec.)
Wave exposure preferences	Exposed, Extremely exposed, Extremely sheltered, Moderately exposed, Sheltered
Other preferences	Rockpool

Additional Information

This biotope is characterized by rockpools with sediment bottoms, in which the presence of sediment and resultant scour restricts algae at the water/sediment interface to sediment tolerant species. However, the biotope describes a range of biotopes from deep, macroalgal dominated pools to shallow pools with only sparse, sediment tolerant, red algae.

The physical characteristics of the rockpool environment are described under 'Seasonal and longer term change' on the 'Ecology' page.

Species composition

Species found especially in this biotope

Rare or scarce species associated with this biotope

-

Additional information

The MNCR database lists 449 species in 58 records of this biotope (JNCC, 1999), although not all species occur in all records of the biotope. The seagrass *Zostera* sp. was recorded from this biotope (see description) and, although no species was given, both *Zostera* species are nationally scarce.

Sensitivity review

Sensitivity characteristics of the habitat and relevant characteristic species

The description of this biotope and information on the characterizing species is taken from Connor *et al.*, (2004). This biotope LR.FLR.Rkp.SwSed describes a spectrum of rockpools from deep, macroalgal dominated pools to shallow sediment filled pools with sparse sediment tolerant red algae. Therefore, while furoids, kelps, and green algae may be characterizing species, their abundance and/or presence is likely to vary between pools, and their loss would not herald a loss of an identifiable biotope, although species diversity would be greatly reduced. The most characteristic feature of this rockpool biotope is the presence of sediment, and hence sediment tolerant red algae. Therefore, the red algae *Furcellaria lumbricalis*, *Polyides rotunda*, *Ahnfeltia plicata* and *Rhodothamniella floridula* have been suggested as characterizing, as they are all examples of sediment tolerant species. However, it should be noted that not all of the above species occur in all records of the biotope, and hence for sensitivity assessment, sediment tolerant algae will be treated as a functional group rather than individually. Grazing by *Littorina littorea* can produce dramatic effects on both the algal assemblage (Lubchenco, 1978, 1983; Robles, 1982; Albrecht, 1998) and habitat structure (Bertness, 1984) of the intertidal zone. For these reasons *Littorina littorea* is also included as a characterizing species within this assessment, as it is the main grazer of the macroalgal community within this biotope. Amphipods are also present within this biotope. However, at the fully marine salinity regime experienced within this biotope the *Littorina littorea* will act as the dominant grazer. In the field *Littorina littorea* is replaced by amphipods as the dominant mesograzer when salinities regularly fall below 12% (Johannesson *et al.*, 1988).

Resilience and recovery rates of habitat

Rockpools which contain this biotope are characterized by the presence of sediment at the bottom. The presence of this sediment requires the characterizing species to be tolerant of chronic disturbance from scour and sediment mobility. As the substratum is relatively mobile, it is likely that the scattered kelps and other larger brown algae that occur in deeper rock pools are usually present as smaller, seasonal recruits that are removed during periods of disturbance and regrow annually. These larger species do not define the biotope and recovery of this element is assessed as 'High' at all levels of impact (resistance is High, Medium, Low or None). Turf forming red algae are the key group characterizing this biotope. Depending on the level of impact, recovery of the turf may occur through repair and regrowth of damaged fronds, regrowth from crustose bases or via recolonization of rock surfaces where all the plant material is removed. Although there are few case studies following recovery some general trends are apparent. The red algae (Rhodophyta) that characterize this biotope have complex life histories and can exhibit distinct morphological stages over their reproductive life history. Alternation occurs between asexual spore producing stages (tetrasporophytes) and male and female plants producing sexually. Life history stages can be morphologically different or very similar. Some species characterizing this biotope possess perennial crustose bases; the thalli can regrow from these crusts where they remain supporting recovery of the biotope (Mathieson & Burns, 1975; Dudgeon & Johnson, 1992). The basal crusts are perennial, tough, resistant stages that prevent other species from occupying the rock surface and allow rapid regeneration and where these remain they provide a significant recovery mechanism. Some species exhibit annual growth and die back patterns. Some temporal variation in abundance and biomass is therefore normal within this biotope. Resistant crustose bases enable the turf of red algae and the crustose corallines to withstand and recover from physical disturbance and scour while preventing the establishment of other species. The spores of

red algae are non-motile (Norton, 1992) and therefore entirely reliant on the hydrographic regime for dispersal. Norton (1992) reviewed dispersal by macroalgae and concluded that dispersal potential is highly variable, recruitment usually occurs on a much more local scale, typically within 10 m of the parent plant. Hence, it is expected that the red algal turf would normally rely on recruitment from local individuals and that recovery of populations via spore settlement, where adults are removed, would be protracted.

Furcellaria lumbricalis is highly fecund, an average sized gametophyte being able to produce approximately 1 million carpospores, or a tetrasporophyte, up to 2 million tetraspores (Austin, 1960a). However, the species grows very slowly compared to other red algae (Bird *et al.*, 1979) and takes a long time to reach maturity. For example, Austin (1960b) reported that, in Wales, *Furcellaria lumbricalis* typically takes 5 years to attain fertility. Bunker *et al.*, (2012) also notes that it can take between 4 – 7 years for this species to reach sexual maturity. This would mean that, following perturbation, recovery to a mature reproductive community would take between 4 - 7 years. Norton (1992) reviewed dispersal by macroalgae and concluded that dispersal potential is highly variable. Spores of *Ulva* sp. have been reported to travel 35km, *Phycodrys rubens* 5km and *Sargassum muticum* up to 1km (Norton, 1992). However, the point is made that reach of the furthest propagule and useful dispersal range are not the same thing and recruitment usually occurs on a much more local scale, typically within 10m of the parent plant. Hence, it is expected that *Furcellaria lumbricalis* would normally only recruit from local populations and hence recovery would be even more protracted in isolated areas. Christensen (1971; taken from Bird *et al.*, 1991) noted that following harvesting of *Furcellaria lumbricalis* forma *aegagropila* in the Baltic Sea, harvestable biomass had not been regained 5 years after the suspension of harvesting. In view of its slow growth, time to maturity and limited dispersal, recoverability of *Furcellaria lumbricalis* is assessed as moderate.

No information was found concerning the longevity of *Ahnfeltia plicata*. However, it is a slow maturing perennial (Dickinson, 1963) and the thallus survives several years without considerable losses (Lüning, 1990). Maggs & Pueschel (1989) reported that mature gametophytes in Nova Scotia varied in size from 3-21 cm, and that 14 months after germination, gametophyte fronds had reached up to 5 cm in length. No definitive information was found concerning age at maturity. However, Maggs & Pueschel (1989) made observations of *Ahnfeltia plicata* from Nova Scotia. Tetrasporophyte crusts matured and released tetraspores after 15 months.

Very little information was found concerning the longevity or the recruitment ability of *Polyides rotunda*. Fletcher & Callow (1992) reported that the spores of *Polyides rotunda* are non-flagellate, meaning the whole distribution process is passive. In general, plankton struggle to re-enter the benthic boundary layer. Therefore, it is likely that successful colonization is achieved under conditions of limited dispersal and/or minimum water current activity. Norton (1992) reported that although spores may travel long distances (e.g. *Ulva* sp. 35 km, *Phycodrys rubens* 5 km), the reach of the furthest propagule does not equal useful dispersal range, and most successful recruitment occurs within 10 m of the parent plants. It is expected, therefore, that recruitment of *Polyides rotunda* would occur from local populations and that establishment and recovery of isolated populations would be patchy and sporadic.

Rhodothamniella floridula is attached to the substratum and is considered unable to re-attach following removal. Recovery of this species therefore depends on the supply of propagules to repopulate surfaces where the population is removed or partly removed. No information was found relating to colonization or recolonization rates of *Rhodothamniella floridula* and some inferences have been made from the distribution of this species and from the life-history traits of

similar species. *Rhodothamniella floridula* is widely distributed around the UK, so that potentially suitable shores can be recolonized from adjacent populations. The species appears to have an annual life-cycle, the sandy turfs produced by this algae are thickest in the winter and become bleached in summer. Red algae are typically highly fecund, but their spores are non-motile (Norton, 1992) and therefore highly reliant on the hydrodynamic regime for dispersal. Kain (1975) reported that after displacement some Rhodophyceae were present after 11 weeks, and after 41 weeks, in June, Rhodophyceae species predominated. However, Stegenga (1978) noted that tetrasporangia of *Rhodothamniella floridula* (as *Rhodochorton floridulum*) germinated in 'rather low numbers'.

Littorina littorea is a very common species of gastropod mollusc which is found throughout the British Isles. It is common on a range of intertidal shore habitats, and can be found from in muddy estuaries and fully marine rocky shores. *Littorina littorea* reproduces annually over an extended period, the egg capsules are shed directly into the sea. Egg release is synchronized with spring tides and occurs on several separate occasions. In estuaries, the population matures earlier in the year and maximum spawning occurs in January (Fish, 1972). A large female (27 mm shell height) may produce up to 100,000 egg capsules per year. Larval settling time or pelagic phase can be up to six weeks conferring high dispersal potential in the water column. Although it is thought that the larvae stay close to the shore during these 6 weeks. The age of maturity in this species is 2 – 3 years, and the life expectancy is upwards of 5 years (Fish & Fish, 1996). As a mobile species, *Littorina littorea* can remove itself from damaging pressures. Although adult immigration is usually an unlikely means of recovery, given this slow crawling, it may be possible within this biotope due to the likelihood of similar rockpools and *Littorina littorea* populations in close proximity.

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), *Lithothamnion* spp. and *Phymatolithon* spp. 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 the encrusting coralline algae, *Lithophyllum incrustans*. Studies by Edyvean & Forde (1987) in populations of *Lithophyllum incrustans* in Pembroke south-west Wales suggest that reproduction occurs on average early in the third year. Reproduction may be sexual or asexual. Populations release spores throughout the year but abundance varies seasonally, with the populations studied in Cullercoats Bay, and Lannacombe Bay (North East and South West England, respectively) producing less spores in the summer. Spore release is initiated by changes in temperature or salinity (see relevant pressure information) at low tide so that spore dispersal is restricted to within the tide pool enhancing local recruitment. Within subtidal biotopes this is not possible and recruitment success may be altered (although this may be compensated for by avoidance of desiccation). Spore survival is extremely low with only a tiny proportion of spores eventually recruiting to the adult population (Edyvean & Ford, 1986). The spores are released from structures on the surface called conceptacles, these are formed annually and subsequently buried by the new layer of growth. Plants can be aged by counting the number of layers of conceptacles. Edyvean & Ford (1984) found that the age structure of populations sampled from Orkney (Scotland) Berwick (northern England) and Devon (England) were similar. Mortality seemed highest in younger year classes and lowest in those individuals which survived past the 10 year mark. In St Mary's Northumberland, the population was dominated by the age 6-7 year classes (Edyvean & Ford, 1984). Growth rates were highest in young plants measured at Pembroke (south-west Wales) with an approximate increase in diameter of plants of 24 mm in year class 0 and 155 mm in year 1 and slowing towards an annual average horizontal growth rate of 3mm/year (Edyvean & Ford, 1987). Some repair of damaged encrusting

coralline occurs through vegetative growth. Chamberlain (1996) observed that although *Lithophyllum incrustans* was quickly affected by oil during the *Sea Empress* spill, recovery occurred within about a year. The oil was found to have destroyed about one third of the thallus thickness but regeneration occurred from thallus filaments below the damaged area. Recolonization by propagules is also an important recovery mechanism, Airoidi (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.

Resilience assessment. Of the characterizing algal species within this biotope the only species with information available on life expectancy and recruitment is *Furcellaira lumbricalis*. Therefore the resilience assessment is made based on this. Although the number of species found within this biotope supplied by Connor *et al.*, (2004) is relatively large, it must be noted that the actual biotope biodiversity is often not as high as this list suggests. The range of rock pools which can contain this biotope vary hugely and all of them will create niches for different species. Therefore when considering a return to previous ecological function after a pressure can vary between biotope. Growth, maturity and longevity vary between the turf forming red algae species. Although some general trends are apparent. Recovery rates, for example, will be greatly influenced by whether the crust stages remain from which the thalli can regrow the characterizing red algae. Biotope resilience is assessed as 'High' where resistance is 'High'. Where resistance is assessed as 'Medium' (loss of <25% of individuals or cover) and the bases remain then recovery is assessed as 'High' based on regrowth from crusts and remaining plants. Where resistance is assessed as 'Medium', 'Low' or 'None', and a high proportion of bases are lost, then recovery will depend on either vegetative regrowth from remaining bases or the supply of propagules from neighbouring populations. Dispersal is limited and propagule supply will be influenced site-specific factors, particularly local water transport. Where resistance is assessed as 'Low' or 'None' then resilience is assessed as 'Medium' (2-10 years) for the red algal turf, and encrusting corallines. Biotope composition may be altered in favour of species with better dispersal ability and higher growth rates but some variation is natural.

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

Hydrological Pressures

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

Rockpools experience considerable variation in temperature on a daily and seasonal basis. The range and extremes of temperature change increasing with shore height but are also dependent on shading, aspect, topography and depth of the pool (Pyefinch, 1943; Ganning, 1971; Daniel &

Boyden, 1975; Goss-Custard *et al.*, 1979; Morris & Taylor, 1983; Huggett & Griffiths, 1986; Metaxas & Scheibling, 1993). For example, reported temperature ranges for mid to low shore pools include annual maxima and minima of 1-25°C and 2-22°C (Morris & Taylor, 1983), a diurnal range of 24°C (day) and 13°C (night) for a mid shore pool (Daniel & Boyden, 1975), and surface water temperature ranges of 14-19.25°C and 15.5-20.75°C in mid shore pools (Pyefinch, 1943). Temperature stratification within pools may result in higher surface temperatures and lower deep water temperatures in sunlight (Daniel & Boyden, 1977) or be reversed due to wind cooling, night or in winter (Naylor & Slinn, 1958; Ganning, 1971; Morris & Taylor, 1983). The temperature range will limit the distribution of sensitive species within the pools, especially normally sublittoral species, e.g. laminarians. For example, the upper limit of *Bifurcaria bifurcata* within rockpools in Roscoff, France was shown to be limited by the summer temperatures where the surface pool water temperatures exceeded 20°C (Kooistra *et al.*, 1989). Therefore, an increase in ambient temperatures has the potential to reduce the abundance or vertical extent of sensitive species within the biotope.

Furcellaria lumbricalis has a wide geographic range, occurring in Europe from northern Norway to the Bay of Biscay. Novaczek & Breeman (1990) recorded that specimens of *Furcellaria lumbricalis* grew well in the laboratory from 0-25°C with optimal growth between 10 and 15°C. Growth ceased at 25°C and 100% mortality resulted after 3 months of exposure to 27°C. Similarly, Bird *et al.* (1979) recorded optimum growth at 15°C and cessation of growth at 25°C with associated necrosis of apical segments.

Ahnfeltia plicata also has a very wide geographic range, occurring from northern Russia to Portugal. The species is therefore likely to be tolerant of higher temperatures than it experiences in Britain and Ireland. Lüning & Freshwater (1988) incubated *Ahnfeltia plicata* from British Columbia at a range of temperatures for 1 week and tested their survivability by ability to photosynthesize at the end of the incubation period. The species survived from the coldest temperature tested (-1.5°C) to 28°C. Total mortality occurred at 30°C. Lüning & Freshwater (1988) suggested that *Ahnfeltia plicata* was therefore amongst the group of most eurythermal heat tolerant algae. Haglund *et al.* (1987) incubated *Ahnfeltia plicata* from the subtidal in Sweden at a range of temperatures and measured photosynthetic rate. There were no significant results, but photosynthetic rate appeared to be optimal at 15°C and decreased either side of this temperature.

Polyides rotunda are found throughout the North Atlantic Ocean and have related populations in the North Pacific. Growth and survival was tested over a temperature range of -5 to 30°C. *Polyides rotunda* tolerated temperatures from -5 to 27°C, grew well from 5 to 25°C, and had a broad optimal range of 10-25°C. Both species tolerated 3 months in darkness at 0°C (Novaczek & Breeman, 1990).

Rhodothamniella floridula has a global distribution (Guiry & Guiry, 2015) it is, over the whole of its range, subject to a wider range of temperatures than experienced in the British Isles. It is therefore expected that increases in temperature, at the pressure benchmarks, will not result in mortality of the species. Dixon & Irvine (1977) observed that the growth of *Rhodothamniella floridula* (as *Audouinella floridula*) is much faster in winter, whilst in the summer the spongy cushion can become bleached or disrupted. Stegenga (1978) found that tetraspores of cultured *Rhodothamniella floridula* (as *Rhodochorton floridulum*) were formed under all combinations of temperatures from 4°C to 16°C at any length of daylight, although they were most abundant at high temperatures and long days.

The median upper lethal temperature limit in laboratory tests on *Littorina saxatilis* and *Littorina*

littorea collected in the summer at Great Cumbrae, Scotland), was approximately 35°C (Davenport & Davenport, 2005). *Semibalanus balanoides* collected from the same shores had similarly high thermal tolerance, with summer collected individuals having a median upper lethal limit of approximately 35°C. In laboratory experiments *Littorina littorea* collected from the Kiel Fjord in Germany and kept in tanks at temperatures 5°C above the seawater temperatures from the collection area (Kiel fjord, Germany) for 5 months (temperatures in laboratory ranged from 13-23°C) did not die although some decreases in shell strength were observed (Landes & Zimmer, 2012).

Chondrus crispus has a wide distribution; it is found extensively throughout Europe and North America. It is also recorded under a number of synonyms from Africa and Asia (Guiry & Guiry, 2015). Spore germination in *Chondrus crispus* appears to be temperature dependent with spores discharged at temperatures of 5°C failing to germinate, although in laboratory culture at 10°C spores were viable all year round (Bhattacharya, 1985). In New Hampshire, USA, *Chondrus crispus* grows abundantly in waters with an annual variation in surface temperature from -1 to +19°C (Mathieson & Burns, 1975). The optimum temperature for growth has been reported as 10-15°C (Fortes & Lüning, 1980), 15 °C (Bird *et al.*, 1979), 15-17°C (Tasende & Fraga, 1999) and 20°C (Simpson & Shacklock, 1979). Above the optimum temperature, growth rate is reported to decline (Bird *et al.*, 1979; Simpson & Shacklock, 1979). Compared to *Chondrus crispus* plants grown at 5°C, plants grown at 20 °C had higher growth rates in terms of length, biomass, surface area, dichotomy and branch production. The differences resulted in growth of morphologically more complex thalli at higher temperatures with more efficient nutrient exchange and light harvesting (Kuebler & Dudgeon, 1996). *Chondrus crispus* plants acclimated to growth at 20°C (vs. 5°C) had higher levels of chlorophyll a and phycobilins, resulting in higher rates of light limited photosynthesis for a given photon flux density (Kuebler & Davison, 1995). Plants grown at 20°C were able to maintain constant rates of light saturated photosynthesis at 30°C for 9 hours. In contrast, in plants acclimated to 5°C, light saturated photosynthetic rates declined rapidly following exposure to 30°C (Kuebler & Davison, 1993). Prince & Kingsbury (1973) reported cessation of growth in *Chondrus crispus* cultures at 26°C, first mortality of spores at 21.1°C and total mortality of spores at 35-40°C, even if exposed for just 1 minute.

Sensitivity assessment. The geographic distribution and laboratory experiments indicate that the key characterizing species would tolerate either an acute or chronic increase in temperature at the pressure benchmark. However, some sub-lethal decreases in photosynthesis may occur where temperatures exceed the optimal. Gamete release may also depend on lower winter temperatures and therefore non-lethal effects on reproduction may occur where optimal temperatures are exceeded. As these effects do not result in mortality, resistance is assessed as 'High' and recovery as 'High' so that the biotope is considered to be 'Not sensitive' to short-term (not greater than a year) changes at the pressure benchmark. However, it should be noted that species may be acclimated to prevailing environmental conditions, so that distribution is only a proxy indicator of sensitivity.

**Temperature decrease
(local)**

High

Q: Medium A: Medium C: Medium

High

Q: Medium A: Medium C: Medium

Not sensitive

Q: Medium A: Medium C: Medium

Rockpools experience considerable variation in temperature on a daily and seasonal basis. The range and extremes of temperature change increasing with shore height but also dependent on shading, aspect, topography and depth of the pool (Pyefinch, 1943; Ganning, 1971; Daniel & Boyden, 1975; Goss-Custard *et al.*, 1979; Morris & Taylor, 1983; Huggett & Griffiths, 1986;

Metaxas & Scheibling, 1993). For example, reported temperature ranges for mid to low shore pools include annual maxima and minima of 1-25 °C and 2-22 °C (Morris & Taylor, 1983), a diurnal range of 24 °C (day) and 13 °C (night) for a mid shore pool (Daniel & Boyden, 1975), and surface water temperature ranges of 14-19.25 °C and 15.5-20.75 °C in mid shore pools (Pyefinch, 1943). Temperature stratification within pools may result in higher surface temperatures and lower deep water temperatures in sunlight (Daniel & Boyden, 1977) or be reversed due to wind cooling, or in winter (Naylor & Slinn, 1958; Ganning, 1971; Morris & Taylor, 1983). Morris & Taylor (1983) reported that the surface of an upper shore was seen to freeze one winter night, although that this was a rare event. Freezing is likely to be rare in mid or low shore pools. The pool is likely to represent a buffer from the extreme cold and frosts experienced by fauna and flora on the emergent rock surface. In addition, few macroalgae were damaged as a result of the severe winter of 1962/63 (Crisp, 1964a). Overall, the range of temperatures routinely experienced by mid to low shore rock pools is greater than the benchmark level. All of the characterizing species within this biotope have broad geographic distributions and are found in colder waters than those around the UK. Populations can acclimate to prevailing conditions which can alter tolerance thresholds and care should therefore be used when interpreting reported tolerances.

Furcellaria lumbricalis occurs in Europe from northern Norway to the Bay of Biscay. Novaczek & Breeman (1990) recorded that specimens of *Furcellaria lumbricalis* grew well in the laboratory from 0-25 °C with optimal growth between 10 and 15 °C. Growth ceased at 25 °C and 100 % mortality resulted after 3 months exposure to 27 °C. Similarly, Bird (1979) recorded optimum growth at 15 °C.

Ahnfeltia plicata occurs from northern Russia to Portugal. Lüning & Freshwater (1988) incubated *Ahnfeltia plicata* from British Columbia at a range of temperatures for 1 week and tested their survivability by ability to photosynthesize at the end of the incubation period. The species survived from the coldest temperature tested (-1.5 °C) to 28 °C. Haglund *et al.* (1987) incubated *Ahnfeltia plicata* from the subtidal in Sweden at a range of temperatures and measured photosynthetic rate. There were no significant results, but photosynthetic rate appeared to be optimal at 15 °C and decreased either side of this temperature.

Growth and survival of the key characterizing species, *Polyides rotunda* was tested over a temperature range of -5 to 30°C. *Polyides rotunda* tolerated temperatures from -5 to 27°C, grew well from 5 to 25°C, and had a broad optimal range of 10-2 °C (Novaczek & Breeman, 1990). This species tolerated three months in darkness at 0°C (Novaczek & Breeman, 1990).

Dixon & Irvine (1977) observed that the growth of *Rhodothamniella floridula* (as *Audouinella floridula*) is much faster in winter, whilst in the summer the spongy cushion can become bleached or disrupted. It is therefore likely that a reduction in temperature will increase the growth rate of the species. *Rhodothamniella floridula* has a global distribution (Guiry & Guiry, 2015, and references therein). However, low temperatures may delay or slow reproduction. Stegenga (1978) found that spores of cultured *Rhodothamniella floridula* (as *Rhodochorton floridulum*) were formed under all combinations of temperatures from 4 °C to 16 °C at any length of daylight, although they were most abundant at high temperatures and long days.

Davenport & Davenport (2005) undertook a series of experiments which found that the median lower lethal temperature tolerances for *Littorina saxatilis* and *Littorina littorea* were -16.4 and -13 °C respectively. In other experiments *Littorina littorea* were able to tolerate temperatures down to -8 °C for 8 days (Murphy, 1983). In colder conditions an active migration may occur down the shore to a zone where exposure time to the air (and hence time in freezing temperatures) is less.

Chondrus crispus has a broad geographical distribution (Guiry & Guiry, 2015) and throughout the range experience wide variation in temperatures (although local populations may be acclimated to the prevailing thermal regime). In New Hampshire, USA, *Chondrus crispus* grows abundantly in waters with an annual variation in surface temperature from -1 to +19 °C (Mathieson & Burns, 1975). The photosynthetic rate of *Chondrus crispus* recovered after 3hrs at -20 °C but not after 6 hrs (Davison *et al.*, 1989). Frond bleaching and declines in photosynthesis and growth also occur in long-term experimental exposure to periodic freezing in *Chondrus crispus* (Dudgeon *et al.*, 1990). Plants from Maine, USA, were frozen at -5 °C for 3 hours a day for 30 days. Photosynthesis was reduced to 55 % of control values, growth rates were reduced and fronds were eventually bleached and fragmented resulting in biomass losses. Additionally, fronds of *Chondrus crispus* which were frozen daily had higher photosynthetic rates following subsequent freezing events than unfrozen controls, indicating that the species is able to acclimate to freezing conditions (Dudgeon *et al.*, 1990). Spore germination in *Chondrus crispus* appears to be temperature dependent with spores discharged at temperatures of 5 °C failing to germinate although in laboratory culture at 10 °C spores were viable all year round (Bhattacharya, 1985). 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.

Sensitivity assessment. The key characterizing species occur over a wide geographical range and can tolerate temperatures below 0 °C. Reduced temperatures may result in suboptimal growth and may affect reproduction. Biotope resistance is assessed as 'High' as these effects do not result in mortality and resilience is assessed as 'High', so that the biotope is not considered to be sensitive to this pressure. However, it should be noted that species may be acclimated to prevailing environmental conditions, so that distribution is only a proxy indicator of sensitivity.

Salinity increase (local)

Medium

Q: Medium A: Medium C: Medium

Medium

Q: High A: Medium C: Medium

Medium

Q: Medium A: Medium C: Medium

High air temperatures cause surface evaporation of water from pools, so that salinity steadily increases. However, Daniel & Boyden (1975) and Morris & Taylor (1983) reported little variability in salinity over one tidal cycle, and Ganning (1971) suggested that changes in salinity were of limited importance. Morris & Taylor (1983) reported an annual maximum salinity of 36.5 ppt in the pools studied on the west coast of Scotland. Goss-Custard *et al.* (1979) recorded salinities of 34.8 and 35.05 ppt in mid-shore pools. Higher shore pools exhibit greater variation and higher extremes of salinity (Pyefinch, 1943; Ganning, 1971) and different communities but mid to low shore pools are unlikely to experience such extremes unless the emergence regime is increased or they are exposed to hypersaline effluents. This biotope occurs in full salinity (Connor *et al.*, 2004) a change above the pressure benchmark is assessed as a change to above 40 ppt.. Local populations will be acclimated to the prevailing conditions even over small spatial distances and gradients and caution should be used in extrapolating sensitivities.

Furcellaria lumbricalis is a euryhaline species which occurs in a wide range of salinity conditions down to 6-8 psu (Bird *et al.*, 1991). In the Kattegat and the Gulf of St Lawrence, it is reported to compete well with other species at salinities ranging from 25-32 psu (see review by Bird *et al.*, 1991). Growth experiments in the laboratory revealed that optimum growth occurred at 20 psu, the species grew well at 10 psu and 30 psu, but that growth declined above 30 psu to negligible levels at 50 psu (Bird *et al.*, 1979). It is expected that an increase in salinity may cause reduced growth and fecundity, but that mortality is unlikely.

Haglund *et al.* (1987) studied photosynthetic rate of *Ahnfeltia plicata* from the subtidal in Sweden and found that, at constant temperature, rate increased up to the maximum salinity tested (33 psu). No specific evidence was found for the salinity tolerance of the key characterizing species *Polyides rotunda*. The *Rhodothamniella floridula* dominated biotope occurs in full salinity conditions. Although no information has been found on survival in hypersaline conditions, the species occurs in rockpools where evaporation may occasionally lead to higher than normal salinities. Young *Littorina littorea* inhabit rock pools where salinity may increase above 35 psu. Thus, these characterizing species may be able to tolerate some increase in salinity.

Although not considered a characterizing species, more information was found on *Chondrus crispus* than the other red algae within this biotope. *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.

Sensitivity assessment. Based on the reported salinity tolerances of the red algae species found within this biotope, it is considered that an increase at the pressure benchmark may lead to some changes in the biological composition of this biotope. The change is likely to reduce species richness and abundance, although some more tolerant species may persist. Resistance (of the biotope) is therefore assessed as 'Medium' and resilience as 'Medium' and the biotope is assessed as 'Medium'.

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

During periods of emersion, high rainfall will reduce pool salinity or create a surface layer of brackish/nearly freshwater for a period. The extremes of salinity experienced will depend on the depth of the pool, shore height and flushing rate, and season. For example, Morris & Taylor (1983) stated that a low salinity layer of 2-10 mm was normal but after one storm the low salinity layer increased in depth, eventually resulting in a homogeneous pool of brackish water. Morris & Taylor (1983) reported an annual salinity range in mid to low shore pools of 26-36.5 ppt. Therefore, decreases in salinity equivalent of a reduction from full to reduced (see benchmark) are likely to be a regular occurrence in rockpool communities. This biotope is present in full salinity (30-35 ppt, Connor *et al.*, 2004), the assessed change at the pressure benchmark is a reduction in salinity to a variable regime (18-35 ppt) or reduced regime (18-30 ppt). Populations can acclimate to local conditions but caution should be used in extrapolating observations from areas where populations are likely to be adapted to the prevailing conditions. A comparative study of salinity tolerances of macroalgae collected from North Zealand in the South Kattegat (Denmark) where salinity is 16 psu. Showed that species generally had a high tolerance (maintained more than half of photosynthetic capacity) to short-term exposure (4 days) to salinities lower than 3.7 (Larsen & Sand-Jensen, 2006). However, tolerances varied between species with *Ahnfeltia plicata*, *Phyllophora pseudoceranoides* and *Chondrus crispus* exhibiting greater tolerance than *Rhodomela confervoides* (Larsen & Sand-Jensen, 2006). The result illustrates that responses to this pressure will vary between species and that a change at the pressure benchmark is likely to alter the

composition of the red algal turf that characterizes the biotope and may alter the biomass and density of more tolerant species.

Furcellaria lumbricalis occurs in the lowest category on the salinity scale (Connor *et al.*, 2004) and therefore probably relatively tolerant of decreases in salinity. The species forms extensive populations in the main basin of the Baltic Sea where salinity is 6-8 psu in the upper 60-70 m and its extension into the Gulfs of Bothnia and Finland is limited by the 4 psu isohaline (see review by Bird *et al.*, 1991).

Ahnfeltia plicata occurs over a very wide range of salinities. The species penetrates almost to the innermost part of Hardanger Fjord in Norway where it experiences very low salinity values and large salinity fluctuations due to the influence of snow melt in spring (Jorde & Klavestad, 1963). *Ahnfeltia plicata* penetrates further than the euryhaline species, *Polyides rotunda*, and probably has a similar salinity tolerance to *Furcellaria lumbricalis*, which is limited only by the 4 psu isohaline (see review by Bird *et al.*, 1991). Haglund *et al.* (1987) studied photosynthetic rate of *Ahnfeltia plicata* from the subtidal in Sweden and found that, at constant temperature, photosynthesis was positively correlated with salinity between 15 and 33 psu. It is likely, therefore, that the benchmark decrease in salinity would not result in mortality, but photosynthesis would not be optimal and so growth and reproduction may be compromised.

No information was found on the effects of reduced salinity on *Rhodothamniella floridula*. However, as this species occurs only in full salinity conditions it is probable that a proportion of the population would die in lower salinities. *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 at the pressure benchmark. *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.

Sensitivity assessment. *Furcellaria lumbricalis* is likely to tolerate a reduction in salinity at the pressure benchmark. The algae species *Ahnfeltia plicata* and *Polyides rotunda*, as well as littorinids are euryhaline and occur over a range of salinities. Some changes in algal composition may occur as a response to a decrease in salinity but may not significantly alter it from the biotope description. Resistance is, therefore, assessed as 'Medium' and resilience as 'Medium' (as bases may not remain to enhance recovery), and biotope sensitivity is judged to be 'Medium'.

Water flow (tidal current) changes (local)

High

Q: Medium A: Medium C: Medium

High

Q: High A: High C: Medium

Not sensitive

Q: Medium A: Medium C: Medium

This biotope occurs across a range of flow speeds, from Strong (1.5 – 3.0 m/s) to areas where water flow is negligible (Connor *et al.*, 2004). As water velocity increases foliose macroalgae 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 in areas that experience a range of flow speeds. Biogenic habitat structures, including the fronds of algae, reduce the effects of water flow on individuals by slowing

and disrupting flow. Boller and Carrington (2006) found that the canopy created by a turf of *Chondrus crispus* reduced drag forces on individual plants by 15-65%. The compact, turf forming growth of the algal species will therefore reduce water flow and the risk of displacement through turbulence and friction. Scour is a key factor structuring this biotope, changes in the flow may increase or decrease sediment transport and associated scour. Reductions in flow may lead to increased deposition of silts and alter the sediment character. An increase in water flow at the pressure benchmark may re-suspend and remove sand particles which are less cohesive than mud particles. The level of impact will depend on site specific hydrodynamic and sediment conditions. Some periodic movement of sediments and changes in coverage is part of the natural temporal variation and periodic disturbances from storms may be more important than water flow in maintaining the character of the biotope, particularly in sheltered areas.

Very little information was available on the effect of water flow on *Furcellaria lumbricalis*, *Ahnfeltia plicata*, and *Polyides rotunda*. However, these species are found in biotopes ranging from strong to negligible. Therefore, only if water flows were to increase above 6 knots (> 3 m/sec) these species may decrease in abundance.

Rhodothamniella floridula is associated occurs in areas where the water flow rate is either moderately strong (0.5-1.5 m/s) or weak (<0.5 m/s) (Connor *et al.*, 2004). Moderate water movement is beneficial to seaweeds as it carries a supply of nutrients and gases to the plants and removes waste products.

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 (3 m/s) may cause snails in less protected locations (e.g. not in crevices etc.) to be continually displaced into unsuitable habitat so that feeding may become sub-optimal. Thus, populations of *Littorina littorea* are likely to reduce. Shell morphology within littorinids varies according to environmental conditions. In sheltered areas shell apertures are small to inhibit predation where *Carcinus maenas* is more prevalent while in exposed areas the foot surface is larger to allow greater attachment and the shell spire is lower to reduce drag (Raffaelli 1982; Crothers, 1992).

Sensitivity assessment. As the biotope can occur in a range of flow speeds, resistance of the biotope to changes in water flow that do not alter the substrata is assessed as 'High' and resilience as 'High' (by default) so that the biotope is assessed as 'Not sensitive'.

Emergence regime changes

Low

Q: High A: Medium C: Medium

Medium

Q: High A: High C: Medium

Medium

Q: High A: Medium C: Medium

An increase in emergence is likely to significantly affect physico-chemical environment of the rockpool and its resident community. An increase in emergence will increase the time that the pool is exposed to fluctuating air temperatures, wind, rain and sunlight, all of which will affect the and temperature, salinity regime within the pool. Lower shore pools will come to resemble mid shore pool communities, with a reduction in sublittoral species and species sensitive to extremes of temperature, for example the laminarians. For example, the upper limit of *Bifurcaria bifurcata* within rockpools in Roscoff, France was shown to be limited by the summer temperatures where the surface pool water temperatures exceeded 20 °C (Kooistra *et al.*, 1989). Mid shore examples of this biotope are likely to be worst affected. High shore pools tend to support communities of temperature tolerant or opportunistic algae, especially green algae such as *Ulva* spp., and temperature and salinity tolerant species as harpacticoid copepods, ostracods, and small

gastropods. A decrease in emergence will reduce the time the pool spends exposed to the air and cut off from the sea. Therefore, the range of temperatures and oxygen levels characteristic of rockpool environments is likely to decrease. Hence the pool communities will come to resemble low shore pools. Low shore pools are characterized by higher abundance of large macroalgae, such as *Halidrys siliquosa*, *Cystoseira* sp. and laminarians, especially deep pools, and a larger diversity of red algae and macrofauna. However, the presence of sediment within the pools will still favour the sand tolerant algae. Therefore, although the community is likely to increase in diversity the biotope is likely to remain.

Both *Furcellaria lumbricalis* and *Ahnfeltia plicata* is found in the lower intertidal to 12m subtidally (Bunker *et al.*, 2012). Therefore, if examples of this biotope in mid shore were to experience an increase in emergence then this species would most probably decline. If there is a decrease in the emergence then these species may be able to increase in their range up the shore.

Polyides rotunda has a wider physical tolerance range than the two previously mentioned algae. Bunker *et al.* (2012) notes that this species occurs from the mid shore down to 24 m depth. This tolerance range means that *Polyides rotunda* may tolerate a decrease or increase in emergence better than other species.

Rhodothamniella floridula occurs predominantly in the littoral and sublittoral to about 5m depth (Dickinson, 1963; Dixon & Irvine, 1997) (as *Rhodochorton floridulum* and *Audouinella floridula* respectively) and is often found in rockpools. Emergence regime is a key factor structuring this (and other) intertidal biotopes. Increased emergence may reduce habitat suitability for the characterizing species *Rhodothamniella floridula* through greater exposure to desiccation and fluctuations in temperature and salinity when emersed (above the surface) and reduced photosynthetic rates when immersed (submerged). Changes in emergence may therefore lead to physical stress followed by species replacement through competition with species more tolerant of the changed conditions.

An increase in emergence that reduced habitat suitability for the grazing littorinids would allow blooms of ephemeral *Ulva* spp. to develop altering the classification of the biotope to LR.FLR.Eph.EphX.

Sensitivity assessment. As emergence is a key factor structuring the distribution of macroalgae on the shore, resistance to a change in emergence (increase or decrease) is assessed as 'Low'.

Recovery is assessed as 'Medium' (following habitat recovery) and sensitivity is therefore assessed as 'Medium'.

Wave exposure changes (local)

High

Q: High A: High C: High

High

Q: High A: High C: Medium

Not sensitive

Q: High A: High C: Medium

This biotope has been recorded from exposed to sheltered wave conditions. The effect of increased wave exposure is probably dependant on the depth of the pool. Sediment is unlikely to remain in shallow pools in wave exposed conditions. In deep pools, increased wave exposure is likely to result in increased scour, resulting in bare rock at the bottom of the pool, especially where cobbles and pebbles are present. Fine sediments and the infauna they support are likely to be lost. However, if the pool is deep enough, the upper levels of the pool are likely to continue to support macroalgae, especially laminarians and erect corallines. For example, an increase in wave exposure from moderately exposed to very exposed is likely to remove the sediment from all but the deepest pools. The occurrence of this biotope across wave exposure categories, was

considered to indicate, by proxy, that biotopes in the middle of the wave exposure range would tolerate either an increase or decrease in significant wave height at the pressure benchmark.

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). *Littorina littorea* regularly have to abandon optimal feeding sites in order to avoid wave-induced dislodgement. This will result in a decreased growth rate (Mouritsen *et al.*, 1999). Increases in wave exposure above the pressure benchmark will probably cause a decrease in population size of *Littorina littorea*.

Sensitivity assessment. An increase in wave action, exceeding the pressure benchmark, may alter the character of the biotope. The characterizing and associated species would probably accrue damage from abrasion and scour. However, at the pressure benchmark the biotope is assessed as 'High', and resilience is also assessed as 'High' by default. The biotope is therefore considered to be 'Not sensitive'.

Chemical Pressures

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

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

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

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

Radionuclide contamination	No evidence (NEv) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	No evidence (NEv) Q: NR A: NR C: NR
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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
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This pressure is **Not assessed**.

De-oxygenation**Medium**

Q: High A: Medium C: Medium

Medium

Q: High A: High C: Medium

Medium

Q: High A: Medium C: Medium

During emergence, rock pools are closed systems and gaseous exchange occurs over the air/water interface. In shallow pools the surface area to volume ratio is likely to be high, whereas in deep pools the ratio is likely to be low. In addition, the oxygen concentration is dependent on the community present. During the day, photosynthesis uses up CO₂ and produces O₂, in excess of respiration. However, at night respiration by flora and fauna deplete oxygen levels. As a result rock pool environments exhibit marked variation in oxygen levels. In summer, rock pools are likely to be supersaturated with oxygen during the day (Pyefinch, 1943). For example, the greatest range of oxygen saturation of 101.7 % occurred in a seaweed dominated, sediment floored pool, which reach over 190 % saturation on some days (Pyefinch, 1943). Daniel & Boyden (1975) noted that a mid-shore, seaweed dominated pool reached 194 % saturation (ca 15 mg O₂/l) but that oxygenation was also marked in shaded pools. A pool with dense fauna exhibited a maximum saturation of 210 % (Pyefinch, 1943). During photosynthesis, algae absorb CO₂ and as concentrations fall, the pH rises. Morris & Taylor (1983) recorded pH values >9 in rock pools on the Isle of Cumbrae. At night, oxygen levels may fall below 100% saturation and pH will decrease as CO₂ levels increase. Morris & Taylor (1983) noted an annual maximum of O₂ concentration of 400-422 mm Hg (ca 23.4-24.7 mg/l) and an annual minimum of 18-38 mm Hg (ca 1-2.2 mg/l) in mid shore pools (containing *Furcellaria lumbricalis*). Daniel & Boyden (1975) reported O₂ depletion at night, with mid to low shore pools reduced to 8-44 % saturation. They noted that the crab *Carcinus maenas* leaves the pools at night, and that other species with the ability to air-breathe could also do so, e.g. limpets, littorinids, and the shanny *Lipophrys pholis*. They also observed that shrimps gathered at the edge of high shore pools at night, presumably to take advantage of the better oxygenated surface layer (Daniel & Boyden, 1975). The range of extremes in oxygen concentration were greater in summer than in winter. On immersion, the rockpool community was exposed to potentially large, sudden fluctuations in oxygen concentrations depending on season and time of day (Morris & Taylor, 1983). Therefore, rockpools communities are probably exposed to variations equivalent to or greater than the benchmark level on a regular basis and tolerant has been recorded.

The effects of reduced oxygenation on the characterizing red algae are not well studied. Lack of oxygen may impair both respiration and photosynthesis (see review by Vidaver, 1972). No evidence is available to make a sensitivity assessment for the key characterizing algal species. However, the presence of these species in rock pools suggests that they can tolerate some short-term oxygen depletion.

Littorina littorea have a high tolerance for low oxygen conditions and can easily survive 3-6 days of anoxia (Storey *et al.*, 2013). In addition, *Littorina littorea* is an air breather when emersed, so can respire during low tide.

Sensitivity assessment. The short-term exposure to de-oxygenation at this benchmark decreases the severity of the pressure. However, the intertidal nature of the rock pools will mean that at low tide, when the pools are cut off, may become even more de-oxygenated. This may increase the level of de-oxygenation and consequently the severity of the pressure. The resistance of this pressure at the benchmark is assessed as 'Medium' as is the resilience. Giving an overall sensitivity 'Medium'.

Nutrient enrichment**High**

Q: High A: Medium C: Medium

High

Q: High A: Medium C: Medium

Not sensitive

Q: High A: Medium C: Medium

This pressure relates to increased levels of nitrogen, phosphorus and silicon in the marine environment compared to background concentrations. The pressure benchmark is set at compliance with Water Framework Directive (WFD) criteria for good status, based on nitrogen concentration (UKTAG, 2014). The nutrient enrichment of a marine environment leads to organisms no longer being limited by the availability of certain nutrients. The consequent changes in ecosystem functions can lead to the progression of eutrophic symptoms (Bricker *et al.*, 2008), changes in species diversity and evenness (Johnston & Roberts, 2009) decreases in dissolved oxygen and uncharacteristic microalgae blooms (Bricker *et al.*, 1999, 2008).

No direct evidence was found to assess this pressure. A slight increase in nutrient levels could be beneficial for barnacles by promoting the growth of phytoplankton levels and therefore increasing zooplankton levels. However, Holt *et al.* (1995) predict that smothering of barnacles by ephemeral green algae is a possibility under eutrophic conditions.

Opportunistic algae, including *Ulva* sp. can't store nutrients in the thallus (unlike larger, long-lived species) and are adapted to efficiently capture and utilise available nutrients in the water column (Pedersen *et al.*, 2009). A large body of field observations and experiments, surveys and laboratory experiments confirm that *Ulva* sp., can utilise high levels of nutrients for growth (Martínez *et al.*, 2012) and that enhanced recruitment (Kraufvelin, 2007) and growth of this genus can occur in enriched areas (Kennison & Fong, 2013; Vaudrey *et al.*, 2010). In areas where nutrient availability is lower either naturally or through management to reduce anthropogenic inputs, *Ulva* sp. may be negatively affected through reduced growth rate and species replacement (Martínez *et al.*, 2012; Vaudrey *et al.*, 2010).

Johnston & Roberts (2009) undertook a review and meta analysis of the effect of contaminants on species richness and evenness in the marine environment. Of the 47 papers reviewed relating to nutrients as a contaminants, over 75% found that it had a negative impact on species diversity, <5% found increased diversity, and the remaining papers finding no detectable effect. Not all of the 47 papers considered the impact of nutrients on intertidal rocky shores. Yet this finding is still relevant as the meta analysis revealed that the effect of marine pollutants on species diversity were 'remarkably consistent' between habitats (Johnston & Roberts, 2009). It was found that any single pollutant reduced species richness by 30-50% within any of the marine habitats considered (Johnston & Roberts, 2009). Throughout their investigation there were only a few examples where species richness was increased due to the anthropogenic introduction of a contaminant. These examples were almost entirely from the introduction of nutrients, either from aquaculture or sewage outfalls. However research into the impacts of nutrient enrichment from these sources on intertidal rocky shores often lead to shores lacking species diversity and the domination by algae with fast growth rates (Abou-Aisha *et al.*, 1995, Archambault *et al.*, 2001, Arévalo *et al.*, 2007, Diez *et al.*, 2003, Littler & Murray, 1975).

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). Nutrient enrichment can also enhance fouling of *Fucus* fronds by biofilms (Olsenz, 2011). Nutrient enriched environments can not only

increase algae abundance, but the abundance of grazing species (Kraufvelin, 2007).

Changes in community composition on intertidal rocky shores can happen rapidly, and fast growing ephemeral species can become established quickly in the presence of higher concentrations of nutrients. The establishment and growth of these species are not controlled by wave exposure (Kraufvelin, 2007). However, even though these fast growing ephemeral species can become well established quickly, healthy communities on intertidal rocky shores can survive long periods of time, and maintain ecological function after these species have become established (Bokn *et al.*, 2002, 2003, Karez *et al.*, 2004, Kraufvelin, 2007, Kraufvelin *et al.*, 2006b).

Sensitivity assessment. A slight increase in nutrients may enhance growth rates but high nutrient concentrations could lead to the overgrowth of the algae by ephemeral green algae and an increase in the number of grazers. However, if the biotope is well established and in a healthy state the biotope could have the potential to persist. The effect of an increase in this pressure to the benchmark level should not have a negative impact on the biotope. Therefore the resistance has been assessed as 'High'. As there will be nothing for the biotope to recover from, the resilience is also 'High'. These two rankings give an overall sensitivity of 'Not Sensitive'.

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

The organic enrichment of a marine environment at this pressure benchmark leads to organisms no longer being limited by the availability of organic carbon. The consequent changes in ecosystem functions can lead to the progression of eutrophic symptoms (Bricker *et al.*, 2008), changes in species diversity and evenness (Johnston & Roberts, 2009) and decreases in dissolved oxygen and uncharacteristic microalgae blooms (Bricker *et al.*, 1999, 2008).

Johnston & Roberts (2009) undertook a review and meta analysis of the effect of contaminants on species richness and evenness in the marine environment. Of the 49 papers reviewed relating to sewage as a contaminant, over 70% found that it had a negative impact on species diversity, <5% found increased diversity, and the remaining papers finding no detectable effect. Not all of the 49 papers considered the impact of sewage on intertidal rocky shores. Yet this finding is still relevant as the meta analysis revealed that the effect of marine pollutants on species diversity were 'remarkably consistent' between habitats (Johnston & Roberts, 2009). It was found that any single pollutant reduced species richness by 30-50% within any of the marine habitats considered (Johnston & Roberts, 2009). Throughout their investigation there were only a few examples where species richness was increased due to the anthropogenic introduction of a contaminant. These examples were almost entirely from the introduction of nutrients, either from aquaculture or sewage outfalls. However, research into the impacts of organic enrichment from these sources on intertidal rocky shores often lead to shores lacking species diversity and the domination by algae with fast growth rates (Abou-Aisha *et al.*, 1995, Archambault *et al.*, 2001, Arévalo *et al.*, 2007, Diez *et al.*, 2003, Littler & Murray, 1975).

Organic 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). Nutrient enrichment can also enhance fouling of fucoid fronds by biofilms (Olsenz, 2011). Nutrient enriched environments can not only

increase algae abundance, but the abundance of grazing species (Kraufvelin, 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.

Changes in community composition on intertidal rocky shores can happen rapidly, and fast growing ephemeral species can become established quickly in the presence of higher concentrations of nutrients. The establishment and growth of these species are not controlled by wave exposure (Kraufvelin, 2007). However, even though these fast growing ephemeral species can become well established quickly, healthy communities on intertidal rocky shores can survive long periods of time, and maintain ecological function after these species have become established (Bokn *et al.*, 2002, 2003, Karez *et al.*, 2004, Kraufvelin, 2007, Kraufvelin *et al.*, 2006b).

Sensitivity assessment. Little empirical evidence was found to support an assessment of this biotope at this benchmark. *Ulva* sp. could easily increase in abundance, and this species could grow over the slower growing characterizing species. However, when organic carbon returns to pre-pressure levels the niche for the characterizing algae would return. If the crust stages remain, from which the thalli of the characterizing red algae can regrow, then recovery could be quick. Therefore, resistance has been assessed as 'High' and resilience has been assessed as 'High'. This gives an overall sensitivity score of 'Not sensitive'.

A Physical Pressures

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

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

Physical change (to another seabed type)	None Q: High A: High C: High	Very Low Q: High A: High C: High	High Q: High A: High C: High
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This biotope is characterized by a rockpool formed from bedrock, within which finer sediment fractions can be found. Removal of the bedrock would remove the attachment surface for the red algal turf that characterizes this biotope, significantly altering the character of the biotope. Artificial hard substratum may also differ in character from natural hard substratum, so that replacement of natural surfaces with artificial may lead to changes in the biotope through changes in species composition, richness and diversity (Green *et al.*, 2012; Firth *et al.*, 2014) or the presence of non-native species (Bulleri & Airoidi, 2005).

Sensitivity assessment. A change to a sedimentary habitat would reduce habitat suitability for this biotope, resistance is assessed as 'None' and resilience as 'Very Low' as the change is considered to be permanent. Sensitivity is therefore assessed as 'High'.

Physical change (to another sediment type)**Low**

Q: Medium A: Medium C: Medium

Very Low

Q: High A: High C: High

High

Q: Medium A: Medium C: Medium

Generally this pressure is considered to be 'Not relevant' to biotopes occurring on bedrock. However, as this biotope is characterized by the bottom of the rock pools having sediment within them, this pressure may have an impact on the health of the biotope. The sand covering and scour is an important factor supporting development and maintenance of this biotope. Removal of sands may allow species with less scour tolerance to colonize the rock pools, altering the character of the biotope. Siltation by finer sediments may have less impact but may lead to subtle changes and changes such as anoxia at the bedrock/sediment interface may lead to removal or damage of holdfasts and bases. A change to coarser gravels, pebbles and cobbles would increase the degree of abrasion where these are mobile and this may also remove the red algal turf.

Sensitivity assessment. A change in the character of the overlying sediment may alter the character of the biotope. Resistance is therefore assessed as 'Low' and resilience is Very low (the pressure is a permanent change), so that the biotope is considered to have 'High' sensitivity to this pressure.

Habitat structure changes - removal of substratum (extraction)

Not relevant (NR)

Q: NR A: NR C: NR

Not relevant (NR)

Q: NR A: NR C: NR

Not relevant (NR)

Q: NR A: NR C: NR

The species characterizing this biotope occur on rock and would be sensitive to the removal of the habitat. However, extraction of rock substratum is considered unlikely and this pressure is considered to be 'Not relevant' to hard substratum habitats.

Abrasion/disturbance of the surface of the substratum or seabed**Medium**

Q: High A: Medium C: Medium

High

Q: High A: High C: Medium

Low

Q: High A: Medium C: Medium

No studies of the effects of trampling or netting on rockpools were found but studies of the effects on emergent algal communities are probably indicative. For example, moderate (50 steps per 0.09 sq. metre) or more trampling on intertidal articulated coralline algal turf in New Zealand reduced turf height by up to 50%, and the weight of sand trapped within the turf to about one third of controls (Schiel & Taylor, 1999). 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 furoid algae and coralline turf species on the New Zealand rocky shore. Low trampling intensity (10 tramples) reduced furoid cover by 25%, while high intensity (200 tramples) reduced furoid cover by over 90%, although over 97% cover returned within 21 months after spring trampling; autumn treatments took longer to recover due to the delay in recruitment. Coralline bases were seen to peel from the rocks (Schiel & Taylor, 1999) due to increased desiccation caused by loss of the algal canopy. Brosnan & Cumrie (1994) demonstrated that foliose species (e.g. furoids and *Mastocarpus papillatus*) were the most susceptible to trampling disturbance, while turf forming species were more resistant. Boalch *et al.* (1974) and Boalch & Jephson (1981) noted a reduction in furoid cover (especially of *Ascophyllum nodosum*) at Wembury, Devon, when compared with the same transects surveyed 43 years previously. They suggested

that the reduction in fucoid cover was due to the large number of visitors and school groups received by the site.

Lagoons form natural shallow water mesocosms and can attract considerable attention from the general public, educational events and scientists alike. Overall, a proportion of the macroalgal community, and the invertebrates it supports are likely to be removed, depending on trampling intensity, and an intolerance of intermediate has been recorded. Resilience is likely to be high (see additional information below) once physical disturbance has stopped. However, it should be noted that ongoing trampling is likely to result in a long-term reduction in the diversity of affected pools.

Sensitivity assessment. The impact of surface abrasion will depend on the footprint, duration and magnitude of the pressure. Based on evidence from the step experiments and the relative robustness of the red algal turf and encrusting corallines, resistance, to a single abrasion event is assessed as 'Medium' (loss of <25% cover/abundance) and recovery as 'High' (as bases are likely to remain). Therefore, the sensitivity is assessed as 'Low'. Resistance and resilience will be lower (and hence sensitivity greater) to abrasion events that exert a greater crushing force and remove the bases.

Penetration or disturbance of the substratum subsurface

Medium

Q: High A: Medium C: Medium

High

Q: High A: High C: Medium

Low

Q: High A: Medium C: Medium

The characterizing species of this biotope occurs on rock which is resistant to subsurface penetration. The assessment for abrasion at the surface only is therefore considered to equally represent sensitivity to this pressure.

Changes in suspended solids (water clarity)

Medium

Q: High A: High C: Medium

High

Q: High A: Medium C: Medium

Low

Q: High A: Medium C: Medium

The presence of sediment in the rockpool environment is a defining feature of this biotope therefore it is potentially sensitive to changes in suspended solids. Siltation and abrasion, which may be associated with changes in suspended solids are assessed separately. The characterizing sediment tolerant red algae are unlikely to be adversely affected by an increase in suspended sediment. However, other macroalgae, and the community they support, are likely to be adversely affected, as shown above (Daly & Mathieson, 1977; Seapy & Littler, 1982). On wave sheltered shores, sediment may accumulate in low to mid shore pools, which again will favour sand tolerant species and infauna. Overall, the biotope is likely to remain but with a decrease in species diversity (for example see Daly & Mathieson, 1977).

A decrease in suspended sediment may result in a decrease in the amount of sediment found in the bottom of pools. This removal is likely to be greatest in shallow pools or on shores with greater wave exposure. The internal topography of individual pools will affect the rate of erosion, the presence of depressions and crevices may retain sediment longer. A decrease in suspended sediment loads may also reduce food availability of suspension feeders within the biotope. Erosion or removal of sediment from the pools would be detrimental for an infauna, and especially sand tolerant algae which are likely to be out-competed by other red algae.

Furcellaria lumbricalis often occurs in relatively turbid waters. Laboratory experiments by Bird *et al.* (1979) revealed that *Furcellaria lumbricalis* was growth saturated at very low light levels (ca

20 μ E/m²/s) compared to other algae such as *Chondrus crispus* (50-60 μ E/m²/s). They suggest that this may be an explanation why *Furcellaria lumbricalis* is able to proliferate in relatively deep and turbid waters. Similarly, in their review, Bird *et al.* (1999) comment that in all studies, saturation and inhibition radiances were low for *Furcellaria lumbricalis* compared to other macroalgae indicating good competitive ability in the attenuated light of deeper or more turbid waters. Increases in turbidity may provide the species with a competitive advantage over other macroalgae.

Ahnfeltia plicata is not likely to be affected directly by an increase in suspended sediment. In general, subtidal red algae are able to exist at relatively low light levels (*Ahnfeltia plicata* typically occurs as an understory alga beneath *Laminaria* sp. (Lüning, 1990) and so is presumably well adapted to growth in low light conditions. *Rhodothamniella floridula* can occur as exposed mats on hard substratum or may occur beneath canopies of *Fucus* spp. which provide shade, indicating that it is tolerant of both high and low light levels and would not be affected by a change in turbidity at the benchmark. Mats of *Rhodothamniella floridula* are found on shores with high levels of sediment and trap sediments between the thalli. This species is therefore considered to be exposed to and tolerant of high levels of suspended sediment and associated scour.

Littorina littorea is found in turbid estuaries where suspended sediment levels are high.

Sensitivity assessment. Changes in suspended solids at the pressure benchmark may lead to changes in the biotope assemblage. However, no empirical evidence was found to assess this pressure. Increases in suspended solids may lead to greater scour damaging plants and reduce light availability impacting growth. Decreased suspended solids may increase the suitability of the biotope for species less tolerant of scour and turbidity that can out-compete the characterizing species. The assessment is based on the characteristic sands remaining in the biotope covering the rocks and limiting the establishment of other competing species, In this scenario the biotope will still be subject to episodes of re-suspension and water transport. As the species present are tolerant of scour and turbidity, biotope resistance is assessed as 'Medium' and resilience as 'High' (as crustose bases are expected to remain to support recovery) sensitivity is therefore assessed as 'Low'.

Smothering and siltation rate changes (light)

Medium

Q: Medium A: High C: Medium

High

Q: High A: Medium C: Medium

Low

Q: Medium A: Medium C: Medium

Seapy & Littler (1982) reported a decrease in macroalgal cover from 47.3 to 37.5% on a Californian rocky shore due to sediment deposition on the mid to lower shore following rain and flooding. *Corallina* sp. and *Pelvetia* sp. were the most affected macroalgal species, while associated red algae were only slightly affected by the resultant scour. Macroinvertebrates declined in cover from 15.8% to 6.5% particularly barnacle species. Daly & Mathieson (1977) examined intertidal zonation on a shore affected by sand scour, and noted that fucoids were reduced to small or young plants, while sand tolerant species such as *Ahnfeltia plicata* dominated on areas affected by sediment. Smothering by 5 cm of sediment is likely to increase scour and be detrimental to the characterizing red algae species. Red algae whose fronds are long enough not to be smothered by 5 cm of sediment could be negatively impacted by the increase in scour. However, all of the characterizing red algae within this biotope are noted for their tolerance to scour and could therefore tolerate an increase in this pressure at the benchmark. Germlings are more likely to be smothered and killed and are inherently most susceptible to this pressure. Indeed, all 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. Sediment deposition can reduce macroalgal recruitment by: (1) reducing the amount of substratum available for attachment of propagules; (2) scour, removing attached juveniles and (3) burial, altering the light and/or the chemical micro-environment (Devinny & Volse, 1978, Eriksson & Johansson, 2003).

The rockpool environment is likely to be more vulnerable to smothering as sediment is likely to accumulate in, and be retained by the rockpool itself. This effectively increases the depth of the sediment layer in the pool. In wave exposed conditions the sediment may be removed, but in sheltered areas it is likely to be retained for longer than indicated by the benchmark. In deep pools, the macroalgae and associated invertebrates are likely to reduce in depth penetration into the pool while sediment tolerant algae increase. In shallow pools, the depth may be further reduced and the macroalgae restricted to sand tolerant species alone. Overall, while the biotope will remain, smothering is likely to reduce the diversity of the pool, exclude grazing littorinids, and smother small epifaunal species such as sponges, bryozoans and ascidians.

Littorina littorea through grazing and bulldozing actions may directly aid the removal of silts and sediments and remove the algal films that may accumulate silts (Bertness, 1984). On a protected New England rocky beach, Bertness (1984) showed how accumulation of sediments due to the removal of the snail *Littorina littorea* changed the character of the habitat. Bertness (1984) observed a decrease in the abundance of organisms characteristic of hard-bottom habitats, such as barnacles and encrusting algae (cited from Airoidi 2003). Chandrasekara & Frid (1998) specifically tested the siltation tolerance of *Littorina littorea*. Burial to 5 cm caused mortality within 24 hours at simulated summer and winter temperatures if the snails could not crawl out of the sediment (Chandrasekara & Frid, 1998). If the sediment is well oxygenated and fluid (as with high water, high silt content) a few snails (1-6 out of 15 in the experiment, depending on temperature, sediment and water content) may be able to move back up through 5 cm of sediment (Chandrasekara & Frid, 1998). Approximately half of the test individuals could not regain the surface from 1 cm of burial except in the most favourable conditions (low temperatures, high water, high silt when a majority (10 out of 15) of the test cohort surfaced. Field observations support the findings that *Littorina littorea* are generally unable to survive smothering. Albrecht & Reise (1994) observed a population of *Littorina littorea* in a sandy bay near the Sylt island in the North Sea. They found that the accretion of mud within *Fucus* strands and subsequent covering of *Littorina* by the sediment resulted in them suffocating and a significant reduction in their abundance.

Sensitivity assessment. Based on the tolerance of the characterizing red algae to sediment inundation and scour, resistance of the characterizing algae to this pressure at the benchmark is likely to be high. However, the littorinids are not tolerant of sediment inundation and an increase in the pressure at the benchmark could reduce the number of these gastropods significantly. By removing this grazer the ecosystem function would change and it is likely that there would be an increase in algal biomass until the grazers returned. The resistance is assessed as 'Medium' and resilience is assessed as 'High' resulting in a 'Low' sensitivity.

Smothering and siltation rate changes (heavy) No evidence (NEv)
Q: NR A: NR C: NR

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

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

No evidence was found to assess this pressure at the benchmark. A deposit at the pressure benchmark would cover all species with a thick layer of fine materials. Species associated with this biotope such as limpets and littorinids would not be able to escape and would likely suffer

mortality (see evidence for light siltation). 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 water currents rather than removed. Mortality will depend on the duration of smothering; where wave action rapidly mobilises and removes fine sediments, survival of the characterizing and associated species may be much greater.

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
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Not assessed.

Electromagnetic changes	No evidence (NEv) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	No evidence (NEv) Q: NR A: NR C: NR
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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
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Species characterizing this habitat do not have hearing perception but vibrations may cause an impact, however no studies exist to support an assessment.

Introduction of light or shading	High Q: High A: High C: High	High Q: High A: High C: Medium	Not sensitive Q: High A: High C: Medium
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In general, subtidal red algae are able to exist at relatively low light levels (Gantt, 1990). Canopy removal experiments in a rocky sub tidal habitat in Nova Scotia, Canada by Schmidt & Scheibling (2007) did not find a shift in understory macroalgal turfs (dominated by *Corallina officinalis*, *Chondrus crispus* and *Mastocarpus stellatus*) to more light-adapted species over 18 months.

Furcellaria lumbricalis often occurs in relatively turbid waters. Laboratory experiments by Bird *et al.* (1979) revealed that *Furcellaria lumbricalis* was growth saturated at very low light levels (ca 20 $\mu\text{E}/\text{m}^2/\text{s}$) compared to other algae such as *Chondrus crispus* (50 - 60 $\mu\text{E}/\text{m}^2/\text{s}$) and *Fucus serratus* (100 $\mu\text{E}/\text{m}^2/\text{s}$). They suggest that this may be an explanation why *Furcellaria lumbricalis* is able to proliferate in relatively deep and turbid waters. Similarly, in their review, Bird *et al.* (1999) comment that in all studies, saturation and inhibition radiance were low for *Furcellaria lumbricalis* compared to other macroalgae indicating good competitive ability in the attenuated light of deeper or more turbid waters.

Ahnfeltia plicata is not likely to be affected directly by an increase in suspended sediment. *Ahnfeltia plicata* typically occurs as an understory alga beneath *Laminaria* sp. (Lüning, 1990) and so is presumably well adapted to growth in low light conditions. An increase in turbidity would reduce the amount of light reaching the understory. Over the course of a year, this may result in mortality of the *Ahnfeltia plicata* individuals at the limit of their depth range.

Rhodothamniella floridula can occur as exposed mats on hard substratum or may occur beneath canopies of *Fucus* spp. which provide shade, indicating that it is tolerant of both high and low light levels. In addition, Stegenga (1978) found that tetraspores of cultured *Rhodothamniella floridula* (as

Rhodochorton floridulum) were formed at any length of daylight, although they were most abundant at high temperatures and long days.

Coralline crusts are shade tolerant algae, often occurring under a macroalgal 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.

The range of *Littorina littorea* extends down into the subtidal (Fish & Fish, 1996) where light attenuation would mean lower light levels than experienced with increased shading in a rock pool. In tank experiments Evans (1965) found that when *Littorina littorea* was without any macroalgae their vertical distribution was affected by directional light. In these experiments when lit from below *Littorina littorea* tend to adopt a level lower than in the control experiment (Evans, 1965).

Sensitivity assessment. As the key characterizing species colonize a broad range of light environments from intertidal to deeper sub tidal and shaded understorey habitats the biotope is considered to have 'High' resistance and, by default, 'High' resilience and therefore is 'Not sensitive' to this pressure.

Barrier to species movement

Not relevant (NR)

Q: NR A: NR C: NR

Not relevant (NR)

Q: NR A: NR C: NR

Not relevant (NR)

Q: NR A: NR C: NR

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

Death or injury by collision

Not relevant (NR)

Q: NR A: NR C: NR

Not relevant (NR)

Q: NR A: NR C: NR

Not relevant (NR)

Q: NR A: NR C: NR

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

Visual disturbance

Not relevant (NR)

Q: NR A: NR C: NR

Not relevant (NR)

Q: NR A: NR C: NR

Not relevant (NR)

Q: NR A: NR C: NR

Not relevant.

Biological Pressures

Resistance

Resilience

Sensitivity

Genetic modification & translocation of indigenous species

Not relevant (NR)

Q: NR A: NR C: NR

Not relevant (NR)

Q: NR A: NR C: NR

Not relevant (NR)

Q: NR A: NR C: NR

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

Introduction or spread of invasive non-indigenous species

Medium

Q: High A: Medium C: Medium

Very Low

Q: High A: High C: Medium

Medium

Q: High A: Medium C: Medium

The high levels of scour in this biotope will limit establishment of all but scour resistant invasive non-indigenous species (INIS). No direct evidence was found for effects of INIS on this biotope. Several non-native species may occur in this biotope. *Codium fragile* subsp. *Tomentosoides* and *Codium fragile* subsp. *Atlanticum* were introduced from mainland Europe and Japan respectively and may out-compete the native *Codium tomentosum* (Eno *et al.*, 1997). The non-native harpoon weed *Asparagopsis armata* was first recorded in Ireland in Galway Bay in 1939 and Britain in 1949 at Lundy in the Bristol Channel, and may come to dominate rockpools (Guiry & Guiry, 2015), although its effect on other species is not known.

Sargassum muticum is a non-native macroalgae spreading around the coasts of Britain and Europe (see Eno *et al.*, 1997) and is often found in low to mid shore rockpools in the intertidal in areas it has colonized. Although, no studies on its effects on rockpool species were found, studies of its effect on shallow sublittoral macroalgae suggest that it can out-compete furoids and kelps. For example, Stæhr *et al.* (2000) reported that an increase in the abundance of *Sargassum muticum* in the Limfjorden (Denmark) from 1990 to 1997 was accompanied by a decrease in the abundance of thick, slow growing macroalgae such as *Saccharina latissima* (studied as *Laminaria saccharina*), *Codium fragile*, *Halidrys siliquosa*, *Fucus vesiculosus*, and *Fucus serratus*, together with other algae such as *Ceramium virgatum* (as *rubrum*) and *Dictyota dichotoma*. In *Sargassum muticum* removal experiments on the coast of Washington State, Britton-Simmonds (2004) concluded that *Sargassum muticum* reduced the abundance of native canopy algae (especially kelps) by 75% and native understory algae by 50% probably as a result of shading. However, Viejo (1999) noted that mobile epifauna (e.g. amphipods, isopods) successfully colonized *Sargassum muticum* which provided additional habitat. Overall, *Sargassum muticum* can successfully invade rockpools, and would probably out-compete resident furoids and kelp species, and some red algae. Although the results of studies of feeding preferences for *Sargassum muticum* over native macroalgae vary, *Littorina littorea* does feed on this species so shoreline colonization by this species would mean that food was still available (Withers *et al.* 1975). *Littorina littorea* also grazes on degraded or stressed *Didemnum vexillum* individuals (Valentine *et al.*, 2007) and *Codium fragile* ssp. *Tomentosoides* (Schiebling *et al.*, 2008), so gains some benefit from the presence of these species.

A number of invasive red algae have been recorded in the UK, from reported habitat preferences *Bonnemaisonia hamifera* does not appear to be present in scoured environments. In North America *Grateloupia turuturu* is a major competitor of *Chondrus crispus*, although *Grateloupia turuturu* is present in the UK, this large foliose species may not be able to colonize this scoured biotope due to the effects of drag and abrasion.

Sensitivity assessment. The presence of sediment is likely to reduce the chance of an invasive non-native species. However, there is a chance that the highly successful non-native *Sargassum muticum* might be able to invade this biotope. The biotope may remain, but with a reduced species richness due to the loss of some species of macroalgae. Resistance is 'Medium' and resilience could also be 'Medium', but assumes removal of *Sargassum muticum* which is unlikely. Hence, a resilience of 'Very low' has been recorded since the biotope is likely to change, although a viable community will remain. Due to the constant risk of new invasive species, the literature for this pressure should be revisited.

Introduction of microbial pathogens **No evidence (NEv)** **Not relevant (NR)** **No evidence (NEv)**
 Q: NR A: NR C: NR Q: NR A: NR C: NR Q: NR A: NR C: NR

Extracts of some red algae show antimicrobial and antifungal activity limiting disease, *Dilsea carnosa* extracts, for example, limit colony extension in *Microsporium canis* and *Trichophyton varrucosum* (Tariq, 1991), providing some protection against pathogens. No information on microbial pathogens in the characterizing red algae species in this biotope can be found.

Parasitism by trematodes may cause sterility in *Littorina littorea*. *Littorina littorea* are also parasitized by the boring polychaete, *Polydora ciliate* and *Cliona sp*, which weakens the shell and increases crab predation (Stefaniak *et al.*, 2005).

Sensitivity assessment. Based on the lack of evidence for the characterizing red algae species within this biotope an assessment of 'No Evidence' has been given.

Removal of target species **Low** **Medium** **Medium**
 Q: High A: High C: Medium Q: High A: High C: Medium Q: High A: High C: Medium

The direct, physical impacts from harvesting are assessed through the abrasion and penetration of the seabed pressures. It is only the biological impact of target species removal which is assessed at this pressure. Several of the characterizing red algae species are subject to harvesting. *Ahnfeltia plicata* is one of the world's principal commercial agarophytes. It is harvested mainly on the Russian coast of the White Sea as a source of high quality, low sulphate agar (Chapman & Chapman, 1980). In Britain and Ireland, however, *Ahnfeltia plicata* does not occur in sufficient quantities to harvest on a commercial scale (Dickinson, 1963). Commercial utilization of *Furcellaria lumbricalis* is based on the gelling properties of its extracted structural polysaccharide, furcellaran (Bird *et al.*, 1991). Extraction of *Furcellaria lumbricalis* was reviewed by Guiry & Blunden (1991).

Plinski & Florczyk (1984) noted that over-exploitation of *Furcellaria lumbricalis* resulted in severe depletion of stocks. However, no commercial harvest as yet occurs in Britain or Ireland. Overall, while rockpools in areas subject to commercial harvesting may be directly affected, most examples of the biotope are unlikely to be affected by commercial harvesting in the UK. However, due to the relative small size of the community, even small scale hand collecting may have a significant effect.

The removal of kelps or brown algae from this biotope is unlikely to the patchy nature of the rock pools which would be deep enough to contain such macroalgae. Edible seaweeds, carrageen (*Chondrus crispus*) and dulse (*Palmaria palmata*) may occur in low densities in the biotope and be harvested locally. As these species are present only in low densities, ecological effects such as the proliferation of algae (from removal of grazers) or other significant changes to the structure and function of the biotope are not predicted to arise from removal.

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 sp.*, on which it preferentially feeds. The community structure within the biotope is likely to be altered but some individuals are likely to remain. 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-

A broad scale 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).

Sensitivity assessment. This biotope is considered to have 'Low' resistance to removal of targeted species by hand-gatherers and 'Medium' resilience and is therefore considered to have a 'Medium' sensitivity to this pressure. However, intense collection of the characterizing species *Littorina littorea* (the species present in high enough quantities within this biotope to be collected commercially) over extended periods of time could lead to the degradation of the biotope and consequent loss of the biotope.

Removal of non-target species

Low

Q: Medium A: Medium C: Medium

Medium

Q: High A: High C: Medium

Medium

Q: Medium A: Medium C: Medium

Direct, physical impacts from harvesting are assessed through the abrasion and penetration of the seabed pressures. The characterizing species *Fucus ceranoides* is a dominant species within this biotope. The dominance of this characterizing species means it could easily be incidentally removed from this biotope as by-catch when other species are being targeted. The loss of this species and other associated species would decrease species richness and negatively impact on the ecosystem function.

Sensitivity assessment. Removal of a large percentage of the characterizing species would alter the character of the biotope. The resistance to removal is 'Low' due to the easy accessibility of the biotopes location and the inability of these species to evade collection. The resilience is 'Medium', with recovery only being able to begin when the harvesting pressure is removed altogether. This gives an overall sensitivity score of 'Medium'.

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