Mytilus edulis, *Fucus serratus* and red seaweeds on moderately exposed lower eulittoral rock

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

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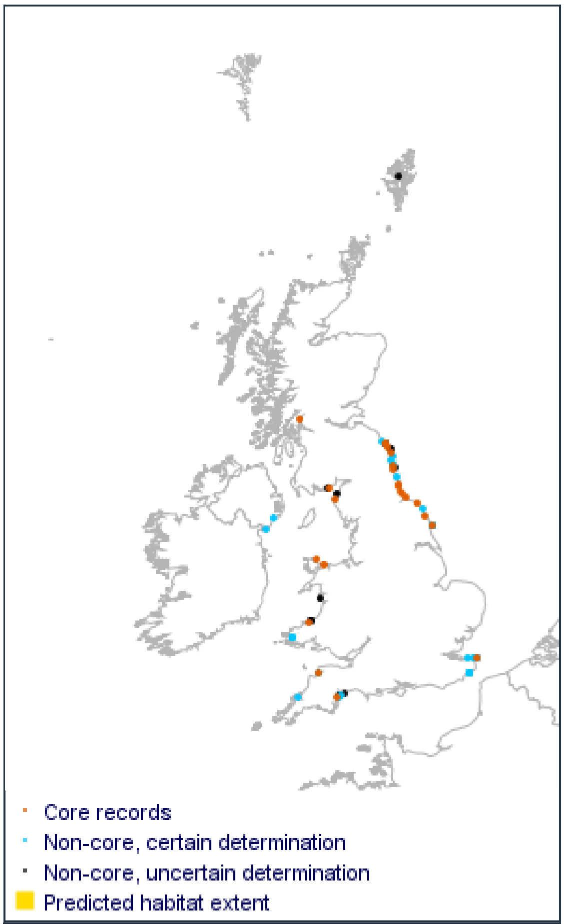

17-09-2018
Biotope distribution data provided by EMODnet Seabed Habitats (www.emodnet-seabedhabitats.eu)

Researched by Dr Heidi Tillin & Kathryn Mainwaring **Refereed by** This information is not refereed.

Summary

UK and Ireland classification

EUNIS 2008	A1.222	*Mytilus edulis*, *Fucus serratus* and red seaweeds on moderately exposed lower eulittoral rock
JNCC 2015	LR.MLR.MusF.MytFR	*Mytilus edulis*, *Fucus serratus* and red seaweeds on moderately exposed lower eulittoral rock
JNCC 2004	LR.MLR.MusF.MytFR	*Mytilus edulis*, *Fucus serratus* and red seaweeds on moderately exposed lower eulittoral rock
1997 Biotope	LR.MLR.MytFR	*Mytilus edulis*, *Fucus serratus* and red seaweeds on moderately exposed lower eulittoral rock

Description

Lower eulittoral moderately exposed bedrock covered by a dense community of large individuals of the mussel *Mytilus edulis*, often with a scarce covering of the wrack *Fucus serratus* and red seaweeds. The red seaweeds may include *Palmaria palmata*, *Mastocarpus stellatus*, *Ceramium* spp., *Audouinella* spp. and *Chondrus crispus*. Ephemeral green seaweeds such as *Ulva intestinalis* and *Ulva lactuca* commonly occur on the shells of the mussels. The barnacle *Semibalanus balanoides* is common on both the mussel valves and on patches of bare rock, where the limpet *Patella vulgata* is also found, often at high abundance. The whelk *Nucella lapillus* and the winkle *Littorina littorea* occur within the mussel bed, as well as the polychaete *Spirobranchus triqueter* and the crab Carcinus maenas. The anemone *Actinia equina* is present in cracks and crevices. These moist areas can be overgrown by coralline crusts (Information from Connor *et al*., 2004; JNCC).

Depth range

Lower shore

Additional information

-

Listed By

- none -

Further information sources

Search on:

G G G JNCC

Sensitivity review

Sensitivity characteristics of the habitat and relevant characteristic species

The biotope description and information on characteristic features is taken from Connor *et al.*, (2004). The sensitivity assessments specifically consider the named species characterizing this biotope: the blue mussel *Mytilus edulis* which occurs as a dense band or large patches and the brown seaweed *Fucus serratus* and red seaweeds which are present but not in high abundances. The red seaweeds may include *Palmaria palmata*, *Mastocarpus stellatus*, *Ceramium* spp., *Audouinella* spp. and *Chondrus crispus and coralline crusts*. The characterizing species define the biotope and provide complex habitats that shelter species and modify environmental conditions. The loss of the characterizing species, in particular *Mytilus edulis*, would lead to reclassification of the biotope, hence the sensitivity assessments focus on these species.

A number of species are associated with this biotope. Ephemeral green seaweeds such as *Ulva intestinalis* and *Ulva lactuca* commonly occur on the shells of the mussels. The barnacle *Semibalanus balanoides* is common on both the mussel valves and on patches of bare rock, where the limpet *Patella vulgata* is also found, often at high abundance. The whelk *Nucella lapillus* and the winkle *Littorina littorea* occur within the mussel clumps. The associated species increase species diversity and support the delivery of ecosystem function and goods and services in general. Predation by *Nucella lapillus* and grazing by *Littorina littorea* and *Patella vulgata* may structure the mussel assemblage and the algae, respectively. Barnacles, *Semibalanus balanoides* interact with the other associated species and hence may structure the assemblage. The sensitivity of the associated species is considered generally in the assessments; particularly where evidence suggests that pressures may leads to changes in abundance that could alter the biotope character.

Resilience and recovery rates of habitat

No evidence for recovery rates of this specific biotope were found. Mainwaring *et al.*, (2014) reviewed evidence for the recovery of *Mytilus edulis* beds rather than dense patches, although the evidence for *Mytilus edulis* beds is considered to have some relevance to this biotope. The algae within the biotope can regrow damaged fronds and blades and may regrow from perennial holdfasts or crustose bases, where these remain. *Mytilus edulis* and other molluscs may be able to repair shells following minor damage but will be more vulnerable to predators and desiccation while healing. Where populations of animals and macroalgae are entirely removed (resistance is none) recovery will require recolonization by propagules. Adults of the mobile species present in the biotope, such as limpets and littorinids may recolonize through adult migration into the habitat from adjacent populations following disturbance or via larval recolonization. In general the animals within the biotope, including *Mytilus edulis*, produce high numbers of pelagic larvae which are widely distributed by water currents, supporting recolonization from surrounding populations following disturbances. Conversely the characterizing red and brown macroalgae generally produce eggs which sink rapidly to the substratum in the vicinity of the adult plants and dispersal distances are short (Dudgeon *et al.*, 2001). Recovery of algal populations may be rapid where adults remain but prolonged where populations are entirely removed.

Mytilus edulis is highly fecund, producing >1,000,000 eggs per spawning event. Larvae stay in the plankton for between 20 days to two months depending on water temperature (Bayne, 1976). In unfavourable conditions they may delay metamorphosis for 6 months (Lane *et al.*, 1985). Larval dispersal depends on the currents and the length of time they spend in the plankton. Settlement occurs in two phases, an initial attachment using their foot (the pediveliger stage) and then a second attachment by the byssus thread before which they may alter their location to a more favourable one (Bayne, 1964). The final settlement often occurs around or between individual mussels of an established population. In areas of high water flow the mussel bed will rely on recruitment from other populations as larvae will be swept away and therefore recovery will depend on recruitment from elsewhere. Larval mortality can be as high as 99% due to adverse environmental conditions, especially temperature, inadequate food supply (fluctuations in phytoplankton populations), inhalation by suspension feeding adult mytilids, difficulty in finding suitable substrata and predation (Lutz & Kennish 1992). After settlement the larvae and juveniles are subject to high levels of predation as well as dislodgement from waves and sand abrasion depending on settlement site.

Recruitment of *Mytilus edulis* is often sporadic, occurring in unpredictable pulses (Seed & Suchanek, 1992), although persistent mussel beds can be maintained by relatively low levels or episodic recruitment (McGrorty *et al*., 1990). A good annual recruitment could result in rapid recovery (Holt *et al*., 1998). However, the unpredictable pattern of recruitment based on environmental conditions could result in recruitment taking much longer. In the northern Wadden Sea, strong year classes (resulting from a good recruitment episode) that lead to rejuvenation of blue mussel beds are rare, and usually follow severe winters, even though mussel spawning and settlement are extended and occur throughout the year (Diederich, 2005). In the List tidal basin (northern Wadden Sea) a mass recruitment of mussels occurred in 1996 but had not been repeated by 2003 (the date of the study), i.e. for seven years (Diederich, 2005).

Height on the shore generally determines lifespan with mussels in the low shore only surviving between 2-3 years due to high predation levels whereas higher up on the shore a wider variety of age classes are found (Seed, 1969). Thiesen (1973) reported that specimens of *Mytilus edulis* could reach 18-24 years of age. As the biotope description refers specifically to ‘large individuals’ (Connor *et al*., 2004) it is considered that the dense patches are relatively long-lived and stable, despite occurring low on the shore, and individuals may have reached as size where predation is limited. Recovery may therefore require a number of years to reach a similar age/size structured *Mytilus edulis* assemblage.

Mainwaring *et al*. (2014) reviewed the evidence for recovery of *Mytilus edulis* beds from disturbance and an earlier study by Seed & Suchanek (1992) reviewed studies on the recovery of ‘gaps’ in *Mytilus* spp. beds. In some long-term studies of *Mytilus californianus* it was observed that gaps could continue to increase in size post disturbance due to wave action and predation (Paine & Levin, 1981; Brosnan & Crumrine, 1994; Smith & Murray, 2005) potentially due to the weakening of the byssus threads leaving them more vulnerable to environmental conditions (Denny 1987). On rocky shores barnacles and fucoids are often quick to colonize the ‘gaps’ created. The presence of macroalgae appears to inhibit recovery whilst the presence of barnacles enhances subsequent mussel recruitment (Seed & Suchanek, 1992). Brosnan & Crumrine (1994) observed little recovery of the congener *Mytilus californianus* in two years after trampling disturbance. Paine & Levin (1981) estimated that recovery times of beds could be between 8-24 years while Seed & Suchaneck (1992) suggested it could take longer-time scales, suggesting that meaningful recovery is unlikely in some areas. It has, however, been suggested that *Mytilus edulis* recovers quicker than other *Mytilus* species (Seed & Suchanek, 1992), which may mean that these predicted recovery rates are too low for *Mytilus edulis.*

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

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

The red algae have complex life histories and exhibit distinct morphological stages over the reproductive life history. Alternation occurs between asexual spore producing stages (tetrasporophytes) and male and female plants producing sexually. Life history stages can be morphologically different or very similar. The tetrasporophyte phase of *Mastocarpus stellatus* is known as the petrocelis and is a flat crust, capable of growing laterally and covering extensive areas. The gametophtyes and tetrasporophytes of *Chondrus crispus* are relatively similar; the holdfasts of individual *Chondrus crispus* can coalesce over time and can form an extensive crust on rock (Taylor *et al*., 1981). Other red algae found within the biotope also have life stages that include prostrate creeping bases e.g. encrusting corallines and *Osmundea pinnatifida* whereas in other species such as *Palmaria palmata*, the thallus or fronds arise from a small discoid holdfast. The basal crusts and crustose tetrasporphytes are perennial, tough, resistant stages that may prevent other species from occupying the rock surface and allow rapid regeneration. They may therefore provide a significant recovery mechanism. Where holdfasts and basal crusts are removed, recovery will depend on recolonization via spores. Norton (1992) reviewed dispersal by macroalgae and concluded that dispersal potential is highly variable, recruitment usually occurs on a much more local scale, typically within 10 m of the parent plant. Hence, it is expected that the algal turf would normally rely on recruitment from local individuals and that recovery of populations via spore settlement, where adults are removed, could be protracted. Minchinton *et* *al*. (1997) documented the recovery of *Chondrus crispus* after a rocky shore in Nova Scotia, Canada, was totally denuded by an ice scouring event. Initial recolonization was dominated by diatoms and ephemeral macroalgae, followed by fucoids and then perennial red seaweeds. After 2 years, *Chondrus crispus* had re-established approximately 50% cover on the lower shore and after 5 years it was the dominant macroalga at this height, with approximately 100% cover.

Coralline crust is a generic term that in UK biotopes refers to nongeniculate (crustose) species from the family Corallinacea that could include *Lithophyllum incrustans* which is noted to form thick crusts in tidepools, especially in the south west (Adey & Adey, 1973), Although ubiquitous in marine coastal systems little is understood about the taxonomy, biology and ecology of this taxa (Littler & Littler, 2013). Throughout the sensitivity assessments the term coralline crust is used to refer to the Corallinacea that occur within the biotope. Due to the lack of evidence for species the assessments are generic, although species specific information is presented where available. A number of papers by Edyvean & Ford (1984a & b; 1986;1987) describe aspects of reproduction and growth of encrusting coralline, *Lithophyllum incrustans*. Studies by Edyvean & Forde (1987) on populations of *Lithophyllum incrustans* in Pembroke south-west Wales suggest that reproduction occurs on average early in the third year. Reproduction may be sexual or asexual. Populations release spores throughout the year but spore abundance varies seasonally. Spore survival is extremely low with only a tiny proportion of spores eventually recruiting to the adult population (Edyvean & Ford, 1986). Edyvean & Ford (1984a) found that the age structure of populations sampled from Orkney (Scotland) Berwick (northern England) and Devon (England) were similar, mortality seemed highest in younger year classes with surviving individuals after the age of 10 years appear relatively long-lived (up to 30 years). In St Mary's Northumberland, the population was dominated by the age 6-7 year classes (Edyvean & Ford, 1984a). Growth rates were highest in young plants measured at Pembroke (south-west Wales) with an approximate increase in diameter of plants of 24 mm in year class 0 and 155 mm in year 1 and slowing towards an annual average horizontal growth rate of 3 mm/year (Edyvean & Ford, 1987). Some repair of damaged encrusting coralline occurs through vegetative growth. Chamberlain (1996) observed that although *Lithophyllum incrustans* was quickly affected by oil during the Sea Empress spill, recovery occurred within about a year. The oil was found to have destroyed about one third of the thallus thickness but regeneration occurred from thallus filaments below the damaged area. Recolonization by propagules is an important recovery mechanism, Airoldi (2000) observed that encrusting coralline algae recruited rapidly on to experimentally cleared subtidal rock surfaces in the Mediterranean Sea, reaching up to 68% cover in 2 months. As encrusting corallines are sensitive to desiccation (Dethier, 1994) it should be noted that these subtidal habitats are probably more favourable for recruitment, growth and survival than intertidal rock pools.

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

Other species that are associated with this biotope, including the limpet *Patella vulgata*, the barnacle *Semibalanus balanoides* and littorinds generally have slower recovery rates due to episodic recruitment and slower growth. Where individuals are removed from a small area, adult limpets and *Littorina saxatilis* may recolonize from surrounding patches of habitat where these are present. The barnacles and limpets and the winkle *Littorina littorea* are common, widespread species that spawn annually producing pelagic larvae that can disperse over long distances. It is therefore likely that adjacent populations will provide high numbers of larvae, although recruitment may be low due to habitat unsuitability and the presence of dense *Ulva* spp. preventing settlement on rock surfaces. *Littorina saxatilis* however brood young and do not have a pelagic life stage, recovery will therefore depend on the presence of adults in close proximity to impacted areas. The larvae of the sea squirt *Ascidiella aspersa* have a short free-swimming planktonic stage. Fertilization to settlement and metamorphosis is estimated to only take about 24 hours at 20 °C (Niermann-Kerkenberg & Hofmann, 1989). The sea squirt *Ascidiella scabra* has a high fecundity and settles readily, probably for an extended period from spring to autumn. Svane (1988) describes it as "an annual ascidian" and demonstrated recruitment onto artificial and scraped natural substrata. It is also likely that *Ascidiella scabra* larvae are attracted by existing populations and settle near to adults (Svane *et al.*, 1987). Fast growth means that a dense cover could be established within about 2 months. However, if mortality occurs at a time when larvae are not being produced, other species may settle and dominate in the freed spaces. The settlement of new colonies of the breadcrumb sponge *Halichondria panicea* is likely to occur within one year with growth rate ranging from ▯ 0.1 to 0.4 cm^2/day. Knowlton & Highsmith (2005) found a rapid response to tissue damage from nudibranch grazing with the sponge recovering within 4 weeks from grazing impacts.

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

Red algae that form turfs, especially *Corallina officinalis*, are often highly resilient to disturbance, and can recover and reach greater abundance compared to prior disturbance conditions (Bulleri *et al.*, 2002; Bertocci *et al.*, 2010). Turf algae can then prevent recovery of fucoids and other species by inhibiting recruitment. Mrowicki *et al.*, (2014) found that limpet and barnacle removal allowed ephemeral and fucoid macroalgae to establish on sheltered and wave exposed shores in Ireland. Experimental studies have shown that limpets and other grazers control the development of macroalgae by consuming microscopic phases (Jenkins *et al.*, 2005) or the adult stages (Davies *et al.*, 2007) and can therefore structure biotopes through feeding preferences (Underwood, 1980; Hawkins & Hartnoll, 1985) Exclusion of grazing limpets, on shores in southern Britain (Swanage and Heybrook), led to the colonization of red algal turfs by *Himanthalia elongata* and *Fucus serratus* within 2 years (Boaventura *et al.*, 2002). MacFarlane (1952) also recorded a shift to a *Corallina officinalis* and encrusting coralline biotope following over raking (for harvesting) of *Chondrus crispus* turf, in these areas gastropods had increased in abundance and prevented the recovery of *Chondrus crispus* by grazing. A change in the abundance of *Patella vulgata* or other grazers could therefore prevent or alter the recovery of this biotope. Opportunistic ephemeral green algae such as *Ulva* sp. can rapidly colonize gaps). These green ephemeral algae are major competitors of *Fucus serratus* for space colonization and nutrient uptake. Blooms of ephemeral algae facilitated by disturbance, particularly where grazers are removed may then slow the development of longer-lived perennial algae, especially fucoids. On the wave exposed and scoured shores that this biotope occurs on, grazing may limit initial settlement of macroalgae but wave action will limit the presence of adults and larger species through breakage and drag effects leading to loss. Limpets and littorinids may enhance barnacle settlement by grazing and removing algae (Hawkins, 1983) or by depositing pedal mucus trails that attract barnacle larvae (Holmes *et al*., 2005). Barnacles and small clumps of *Mytilus edulis* may enhance survival of small limpets by moderating environmental stresses but they may also have negative effects on recruitment by occupying space and by limiting access to grazing areas (Lewis & Bowman (1975).

Resilience assessment. The evidence for recovery rates of *Mytilus edulis* beds from different levels of impact is very limited and whether these rates are similar, or not, between different types of biotope is largely unclear. Recovery rates are clearly determined by a range of factors such as degree of impact, season of impact, larval supply and local environmental factors including hydrodynamics so that confidence in the applicability of generic assessments is 'Low'.
Overall, *Mytilus* spp. populations are considered to have a strong ability to recover from environmental disturbance (Holt *et al*., 1998; Seed & Suchaneck, 1992). A good annual recruitment may allow a bed to recovery rapidly, though this cannot always be guaranteed within a certain time-scale due to the episodic nature of *Mytilus edulis* recruitment (Lutz & Kennish, 1992; Seed & Suchanek, 1992) and the influence of site-specific variables. The sensitivity assessments for this biotope have adopted the rates of *Mytilus edulis* recovery used by Mainwaring *et al*. (2014), as this species and the dense bands it forms may be the slowest element to recover. Where biotope resistance is 'High' then there is no effect to recover from and resilience should be assessed as 'High'. Dense patches of *Mytilus edulis* are considered to have 'Medium' resilience (2 -10 years) to represent the potential for recovery within a few years where a proportion of the bed remains ('Medium' or 'Low' resistance). Resilience of *Mytilus edulis* is assessed as 'Low' (over 10 years) for all biotopes where resistance is assessed as 'None', as recovery is dependent on recruitment from other areas and recruitment can be sporadic and the biotope may come to be dominated by silt tolerant brown and red algae preventing recruitment. Due to the variation in recovery rates reported in the literature, while the evidence for resilience is of 'High' quality and 'High' applicability (for recovery from the same pressures or otherwise assessed as 'Low'), the degree of concordance is 'Medium'.

The characterizing species *Fucus serratus* and the turf forming red algae are considered to have higher potential recovery rates than *Mytilus edulis.* Recovery rates of red algae will be greatly influenced by whether the crust or holdfasts remain from which the thalli can regrow. Where the bases remain and resistance is assessed as 'Medium' (loss of <25% of individuals or cover) then recovery is assessed as 'High' based on regrowth from crusts and remaining plants. Where resistance is assessed as 'Low' or 'None' and a high proportion of bases are lost then recovery may be more protracted. Based on recovery from ice scour (Minchinton *et al*., 1997) recovery is assessed as 'Medium' (2-10 years). As recovery where turfs are removed over large areas will depend on the supply of propagules from neighbouring populations and as dispersal is limited the recovery will depend on the supply of propagules which will be influenced site-specific factors, particularly local water transport.

Caveats regarding possible state shifts where species are removed should be considered when applying sensitivity assessments. Identifying tipping points for shifts to alternate stable states is problematic, therefore although the recovery rates based on examples and life history traits are used in the assessments these may underestimate or overestimate recovery time, which will be influenced by pressure and site-specific factors. If specimens of *Fucus serratus* remain in small quantities it is likely that re-growth will occur rapidly due to efficient fertilization rates and recruitment over short distances. Recovery is likely to occur within two years resulting in a 'High' resilience score. However, if the population is removed (resistance is 'None'), recovery may take longer, perhaps up to 10 years so the resilience would be scored as 'Medium'.

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

Hydrological Pressures

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

Species found in the intertidal are exposed to extremes of high and low air temperatures during periods of emersion. They must also be able to cope with sharp temperature fluctuations over a short period of time during the tidal cycle. In winter, air temperatures are colder than the sea, conversely in summer air temperatures are much warmer than the sea. Species that occur in this intertidal biotope are therefore generally adapted to tolerate a range of temperatures, although the timing of site-specific factors such as low tides will influence local acclimation. For intertidal species increased temperatures may also result in desiccation when exposed (see changes in emergence pressure). Local populations may be acclimated to the prevailing temperature regime and may therefore exhibit different tolerances to other populations subject to different conditions and therefore caution should be used when inferring tolerances from populations in different regions.

Mytilus edulis is a eurytopic species found in a wide temperature range from mild, subtropical regions to areas which frequently experience freezing conditions and are vulnerable to ice scour (Seed & Suchanek,1992). In British waters 29°C was recorded as the upper sustained thermal tolerance limit for *Mytilus edulis* (Read & Cumming, 1967; Almada-Villela, *et al*., 1982), although it is thought that European mussels will rarely experience temperatures above 25°C (Seed & Suchanek, 1992). Tsuchiya (1983) documented the mass mortality of *Mytilus edulis* in in Mutsu Bay, northern Japan in August 1981, due to air temperatures of 34°C that resulted in mussel tissue temperatures in excess of 40°C. In one hour, 50% of the *Mytilus edulis* from the upper 75% of the shore had died. It could not be concluded from this study whether the mortality was due to high temperatures, desiccation or a combination of the two. Lethal water temperatures appear to vary between areas (Tsuchiya, 1983) although it appears that their tolerance at certain temperatures vary, depending on the temperature range to which the individuals are acclimatised (Kittner & Riisgaard, 2005). After acclimation of individuals of *M. edulis* to 18°C, Kittner & Riisgaard (2005) observed that the filtrations rates were at their maximum between 8.3 and 20°C and below this at 6°C the mussels closed their valves. However, after being acclimated at 11°C for five days, the mussels maintained the high filtration rates down to 4°C. Hence, given time, mussels can acclimatise and shifting their temperature tolerance. Filtration in *Mytilus edulis* was observed to continue down to -1°C, with high absorption efficiencies (53-81%) (Loo, 1992).

At the upper range of a mussels tolerance limit, heat shock proteins are produced, indicating high stress levels (Jones *et al*., 2010). After a single day at 30°C, the heat shock proteins were still present over 14 days later, although at a reduced level. In shallow lagoons mortality began in late July at the end of a major spawning event when temperatures peaked at >20°C (Myrand *et al*., 2000). These mussels had a low energetic content post spawning and had stopped shell growth. It is likely that the high temperatures caused mortality due to the reduced condition of the mussels post spawning (Myrand *et al*., 2000). Gamete production does not, however, appear to be affected by temperature (Suchanek, 1985).

Temperature changes may also lead to indirect effects For example, an increase in temperature increases the mussels' susceptibility to pathogens (*Vibrio tubiashii*) in the presence of relatively low concentrations of copper (Parry & Pipe, 2004). Increased temperatures may also allow for range expansion of parasites or pathogens which will have a negative impact upon the health of the mussels if they become infected.

Power stations have the potential to cause an increase in sea temperature of up to 15°C (Cole *et al*., 1999), although this impact will be localised. However, as mussels are of the most damaging biofouling organisms on water outlets of power stations, they are clearly not adversely affected (Whitehouse *et al*., 1985; Thompson *et al*., 2000).

Most fucoids are cold-temperate species (Lüning, 1984), and temperatures above 20 °C are generally considered unsuitable for these algae (Zou *et al*., 2012). The effect of high temperature stress on photosynthesis in brown algae is related to inactivation of enzymes and the induction of reactive oxygen species (ROS), leading to photoinhibition (Suzuki & Mittler, 2006). Growth rates of adult brown macroalgae may be affected by temperature through the increase in metabolic rates (Nygard & Dring, 2008). However, *Fucus serratus* is found along the Atlantic coast of Europe from Svalbard to Portugal and on the shores of north-east America. The seaweed is thus well within its thermal range in the British Isles. Nielsen *et al*. (2014) found no negative effects on growth rates of adult *Fucus serratus* to water temperatures of 22 °C (based on a laboratory experiment with specimen collected from Firth of Forth, Scotland) and Arrontes (1993) observed that *Fucus serratus* survived in laboratory experiments for 1 week at 25 °C. Nielsen *et al*. (2014) did, however, report that germlings were negatively affected by increased temperature indicating that early life stages are more vulnerable than mature algae to this pressure.

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

In an exceptionally hot summer (1983, with an increase of between 4.8 and 8.5 °C), Hawkins & Hartnoll (1985) observed no temperature bleaching of adult *Himanthalia elongata* (although some buttons were bleached) or other canopy forming species. However, understorey red algae showed more signs of damage with bleached *Corallina officinalis* observed around the edges of pools due to dessication. Occasional damaged specimens of *Palmaria palmata*, *Osmundea pinnatifida* and *Mastocarpus stellatus* were observed. *Palmaria palmata* does well in low temperatures, with an optimum between 6 and 15°C, consistent with a distribution in northern temperate and arctic waters. This species is also found in warmer temperate waters as far south as Portugal in Europe and with localized large populations in northern Spain (Garbary *et al*., 2012 and references therein). Temperatures at or above 15 °C may induce physiological stress (Werner & Dring, 2011; Morgan *et al*., 1980). In tank cultures of *Palmaria palmata* at 20°C and above, all plants were dead within a week (Morgan *et al*., 1980). Populations may be acclimated to typical conditions but it is likely that *Palmaria palmata* may be bleached or damaged by higher than usual temperatures.

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

Limpets, *Patella vulgata* and littorinids also occur within this biotope. Laboratory studies suggest that adults of these species can tolerate temperature increases. The median upper lethal temperature limit in laboratory tests on *Littorina littorea*, *Littorina saxatilis* was approximately 35 °C (Davenport & Davenport, 2005). *Patella vulgata* can also tolerate high temperatures. The body temperature of *Patella vulgata* can exceed 36°C in the field, (Davies, 1970); adults become non-responsive at 37-38 C and die at temperatures of 42 °C (Evans, 1948). *Semibalanus balanoides* and *Patella vulgata* are 'northern' with their range extending from Portugal or Northern Spain to the Arctic circle. Populations in the southern part of England are therefore relatively close to the southern edge of their geographic range. Reproductive and recruitment success in both species is linked to temperature and long-term changes in temperature (exceeding the duration of the pressure benchmark) may to lead to replacement by the warm water species *Chthamalus montagui* and *Chthamalus stellatus* (Southward *et al*., 1995). In Northern Portugal warming seas appear to be linked to a shortening of the reproductive period and the lack of multiple spawning events in *Patella vulgata* and other northern species (Ribeiro *et al*., 2009).

Sensitivity assessment. Based on the wide range of temperature tolerance of *Mytilus edulis* and its limited effect on its physiology, it is concluded that the acute and chronic changes described by the benchmarks of 2-5°C would have limited effect. *Fucus seratuss* are found in the middle of their natural temperature range in the British Isles and are therefore not likely to be affected by an increase in temperature at the pressure benchmark. An increase in acute or chronic temperature above average British and Irish temperatures is not likely to have a detrimental effect of *Fucus serratus* and associated communities, based on global distribution. However, it should be noted that phenotypic plasticity will influence the tolerance of individual population. Some of the understorey of red algae, such as *Palmaria palmata* may be lost during acute temperature increases if these occur in the summer when plants are already close to the limit of thermal tolerances. Biotope resistance is assessed as 'High' and resilience as 'High' and the biotope is considered to be 'Not sensitive'. It should be noted that the timing of acute and chronic increases would alter the degree of impact and hence sensitivity. An acute change occurring on the hottest days of the year and exceeding thermal tolerances may lead to mortality. Sensitivity of *Patella vulgata* and *Semibalanus balanoides* to longer-term, broad-scale perturbations would potentially be greater due to effects on reproduction.

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

Many intertidal species are tolerant of freezing conditions as they are exposed to extremes of low air temperatures during periods of emersion. They must also be able to cope with sharp temperature fluctuations over a short period of time during the tidal cycle. In winter air temperatures are colder than the sea, conversely in summer air temperatures are much warmer than the sea. Species that occur in the intertidal are therefore generally adapted to tolerate a range of temperatures, with the width of the thermal niche positively correlated with the height of the shore (Davenport & Davenport, 2005). Local populations may be acclimated to the prevailing temperature regime and may therefore exhibit different tolerances to other populations subject to different salinity conditions and therefore caution should be used when inferring tolerances from populations in different regions. *Mytilus edulis* is a eurytopic species found in a wide temperature range from mild, subtropical regions to areas which frequently experience freezing conditions and are vulnerable to ice scour (Seed & Suchanek, 1992).

The lower lethal limit of *Mytilus edulis* depends on the length of time exposed to a low temperature and the frequency of exposure (Bourget, 1983). Williams (1970) observed that *Mytilus edulis* tolerated a tissue temperature as low as -10 °C. In a laboratory experiment, Bourget (1983) showed that the median lethal temperature for 24 hour of exposure in *Mytilus edulis* was -16 °C for large mussels (>3cm) and -12.5 °C for juveniles (<1.5cm). However, when exposed to reduced temperatures for only 16 hours, the median lethal temperature of large mussels decreased to -20 °C. It was also reported that mussels exposed to sub lethal temperatures cyclically, e.g. -8 °C every 12.4 hours for 3-4 days, suffered significant damage likely to lead to death (Bourget, 1983), which suggested that while *Mytilus edulis* could tolerate occasional sharp frost events it was not likely to survive prolonged periods of very low temperatures. During the cold winter of 1962/63, *Mytilus edulis* was reported to have experienced relatively few effects with only 30% mortality being recorded from the south east coast of England (Whitstable area) and only about 2% mortality was reported from Rhosilli in South Wales (Crisp, 1964). Crisp (1964) also noted that the mortality was mainly from predation on the individuals that were weakened by the low temperatures rather than the temperature itself. It is thought that the use of nucleating agents in the haemolymph and the maintenance of a high osmotic concentration in the mantle fluid during periods of winter isolation allows *Mytilus edulis* to tolerate such low temperatures (Aunaas *et al*., 1988).

Shell growth is not expected to be majorly influenced by low temperatures. Bayne (1976) demonstrated that between 10-20 °C water temperature had little effect on scope for growth, similar to the findings of Page & Hubbard (1987), who found that a temperature range of 10-18 °C did not influence growth rate. In addition, Loo (1992) recorded growth rates of up to 0.7 % at temperatures as low as -1 °C, with an excess of seston, a rate higher than the same author recorded in mussel culture in Sweden (Loo & Rosenberg, 1983). They concluded that food availability was more of a limiting factor to growth than temperature (Loo, 1992).

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

Under extremely low temperatures, components of the community demonstrate tolerance. Lüning (1990) suggested that most littoral algal species were tolerant of cold and freezing. The associated species *Mastocarpus stellatus* has a broad geographical distribution (Guiry & Guiry, 2015) and throughout the range experience wide variation in temperatures (although local populations may be acclimated to the prevailing thermal regime). The photosynthetic rate of *Mastocarpus stellatus* higher on the shore fully recovered from 24 hrs at -20 °C (Dudgeon *et al*.

(1989). Photosynthesis in *Mastocarpus stellatus* also recovered quickly after experimental freezing (Dudgeon *et al*., 1989, 1995). *Palmaria palmata* does well in low temperatures, with an optimum between 6 and 15°C, consistent with a distribution in northern temperate and arctic waters. In the laboratory, plants only became fertile if left at temperatures between 5-7 °C with a short light period (Van der Meer, 1979). Acute or chronic changes in temperature below 5°C may therefore reduce reproductive success although reproduction and vegetative growth in warmer months should compensate for any reduction in output. The green algae, *Ulva* spp. are eurytopic, found in a wide temperature range and in areas which frequently experience freezing conditions and are vulnerable to ice scour (Seed & Suchanek 1992).

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

Sensitivity assessment. Based on the wide range of temperature tolerance of *Mytilus edulis* and its limited effect on its physiology, it is concluded that the acute and chronic changes described by the benchmarks of 2-5°C would have limited effect. A decrease in acute or chronic temperature above average British and Irish temperatures is not likely to have a detrimental effect of *Fucus serratus* and associated communities, based on global distribution. However, it should be noted that phenotypic plasticity will influence the tolerance of individual population. Based on the characterizing and associated species, this biotope is considered to have ‘High’ resistance and ‘High resilience (by default) to this pressure and is therefore considered to be ‘Not sensitive’. The timing of changes and seasonal weather could result in greater impacts on species. An acute decrease in temperature coinciding with unusually low winter temperatures may exceed thermal tolerances and lead to mortalities of the associated species although this would not alter the character of the biotope.

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

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

Mytilus edulis is found in a wide range of salinities from variable salinity areas (18-35 ppt) such as estuaries and intertidal areas, to areas of more constant salinity (30-35 ppt) in the sublittoral (Connor *et al*., 2004). Furthermore, mussels in rock pools are likely to experience hypersaline conditions on hot days. Newell (1979) recorded salinities as high as 42 psu in intertidal rock pools, suggesting that *Mytilus edulis* can tolerate high salinities. In the laboratory, *Semibalanus balanoides* was found to tolerate salinities between 12 and 50 psu (Foster, 1970). Young *Littorina littorea* inhabit rock pools where salinity may increase above 35psu. Thus, the associated animal species may also be able to tolerate an increase in salinity.

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

Tolerances of the red algae vary between species. *Chondrus crispus* is found in a range of salinities across its range and has been reported from sites with yearly salinity range 0-10 psu and 10-35 psu (Lindgren & Åberg 1996) and sites from an average of 26-32 psu. However, at different salinities the ratio between the abundance of the tetrasporophyte phase and the gametophyte alters (Guidone & Grace, 2010). Mathieson & Burns (1971) recorded maximum photosynthesis of *Chondrus crispus* in culture at 24 psu, but rates were comparable at 8, 16 and 32 psu. Photosynthesis continued up to 60 psu. Bird *et al.* (1979) recorded growth of Canadian *Chondrus crispus* in culture between 10 and 50 psu, with a maximum at 30 psu. *Chondrus crispus* would therefore appear to be euryhaline and tolerant of a range of salinities. Laboratory experiments have defined the upper and lethal lower limits for *Palmaria palmata* as 15 psu and 50 psu, (Karsten *et al*., 2003) with optimal salinity defined as 23-34 psu (Robbins, 1978).

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

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

Sensitivity assessment. Little direct evidence was found to assess sensitivity to this pressure.

Although some increases in salinity may be tolerated by the associated species present these are generally short-term and mitigated during tidal inundation. This biotope is considered, based on distribution of *Mytilus edulis*, *Fucus serratus* and the associated red algal species on the mid to lower shore to be sensitive to a persistent increase in salinity to > 40 ppt. Resistance is therefore assessed as 'Low' and recovery as 'Medium' (following restoration of usual salinity). Sensitivity is therefore assessed as 'Medium'.

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

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

Mytilus edulis is found in a wide range of salinities from variable salinity areas (18-35ppt) such as estuaries and intertidal areas, to areas of more constant salinity (30-35ppt) in the sublittoral (Connor *et al.*, 2004). In addition, *Mytilus edulis* thrives in brackish lagoons and estuaries, although, this is probably due to the abundance of food in these environments rather than the salinity (Seed & Suchanek, 1992). Furthermore, mussels in rock pools are likely to experience hypersaline conditions on hot days. Newell (1979) recorded salinities as high as 42 psu in intertidal rock pools, suggesting that *Mytilus edulis* can tolerate high salinities. Also, *Mytilus edulis* was recorded to grow in a dwarf form in the Baltic sea where the average salinity was 6.5psu (Riisgård *et al.*, 2013).

Mytilus edulis is an osmoconformer and maintains its tissue fluids iso-osmotic (equal ionic strength) with the surrounding medium by mobilisation and adjustment of the tissue fluid concentration of free amino acids (e.g. taurine, glycine and alanine) (Bayne, 1976; Newell, 1989). But mobilizing amino acids may result in loss of protein, increased nitrogen excretion and reduced growth. However, Koehn (1983) and Koehn & Hilbish (1987) reported a genetic basis to adaptation to salinity. *Mytilus edulis* exhibits a defined behavioural response to reducing salinity, initially only closing its siphons to maintain the salinity of the water in its mantle cavity, which allows some gaseous exchange and therefore maintains aerobic metabolism for longer. If the salinity continues to fall the valves close tightly (Davenport, 1979; Rankin & Davenport, 1981). In the long-term (weeks), *Mytilus edulis* can acclimate to lower salinities (Almada-Villela, 1984; Seed & Suchanek 1992; Holt *et al.*, 1998). Almada-Villela (1984) reported that the growth rate of individuals exposed to only 13 psu reduced to almost zero but had recovered to over 80% of control animals within one month. Observed differences in growth are due to physiological and/or genetic adaptation to salinity.

Decreased salinity has physiological effects on *Mytilus edulis*; decreasing the heart rate (Bahmet *et al.*, 2005), reducing filtration rates (Riisgård *et al.*, 2013), reducing growth rate (Gruffydd *et al.*, 1984) and reducing the immune function (Bussell *et al.*, 2008). Both Bahmet *et al.*, (2005); Riisgård *et al.*, (2013) noted that filtration and heart rates return to normal within a number of days acclimation or a return to the original salinity. However, Riisgard *et al.*, (2013) did observe that mussels from an average of 17 psu found it harder to acclimate between the salinity extremes than those from an average of 6.5 psu. This observation may mean that mussels in a variable/ lower salinity environment are more able to tolerate change than those found at fully marine salinities. A sharp salinity change also induces a behavioural response to close the shell (Riisgård *et al*., 2012) to maintain the salinity within the mantle cavity. In extreme low salinities, e.g. resulting from storm runoff, large numbers of mussels may be killed (Keith Hiscock pers comm.). However, Bailey *et al*., (1996) observed very few mortalities when exposing *Mytilus edulis* to a range of salinities as low as 0ppt for two weeks at a range of temperatures. It was also noted that there was a fast recovery rate.

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

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

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

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

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

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

Sensitivity assessment. Most of the literature found on this topic considered short-term (days to weeks) impacts of changes to salinity whilst the benchmark refers to a change for one year. *Mytilus edulis* is capable of acclimation to changes in salinity and is found in salinities as low as 4-5psu (Riisgård *et al*., 2013), it is, therefore, likely to be able to acclimate to a decrease in salinity at the pressure benchmark. *Mytilus edulis* is judged ‘High’ resistance to a decrease in salinity (at the pressure benchmark) and ‘High’ resilience (no impact to recover from). A reduction in salinity at the level benchmark (e.g. from 'Full' to 'Reduced' for one year) could have beneficial effects *on Fucus serratus* as growth rates are maximal below full saline conditions. Prolonged reduction in salinity, e.g. from full to reduced, is likely to reduce the species richness of the biotope due to loss of less tolerant red algae and some intolerant invertebrates. The sensitivity of *Palmaria palmata* for example, appears to be greater than some other turf-forming species such as *Chondrus crispus*. A change in salinity at the pressure benchmark is considered to lead to some mortality of *Palmaria palmata* although the tolerance threshold reported in laboratory and cultivation studies is slightly lower than the assessed benchmark. However, the dominant species will probably survive and the integrity of the biotope is likely to be little affected. It is considered that the benchmark decrease in salinity (from full to variable) would not result in mortality of the characterizing species in biotopes that were previously fully marine. Biotope resistance is therefore assessed as 'High' and resilience as 'High', based on no effect to recover from.

Water flow (tidal current) changes (local)	Medium Q: High A: High C: Medium	Medium Q: High A: Low C: Medium	Medium Q: High A: Low C: Medium

Blue mussels are active suspension feeders generating currents by beating cilia and are therefore not entirely dependent on water flow to supply food (organic particulates and phytoplankton). Therefore, they can survive in very sheltered areas, but water flow (due to tides, currents or wave action) can enhance the supply of food, carried from outside the area or resuspended into the water column. The evidence for *Mytilus edulis* sensitivity to changes in water flow was reviewed by Mainwaring *et al*., (2014). Individuals attached to solid substrata (rock) as in this biotope are likely to display more resistance than individuals attached to boulders, cobbles or sediment.

The growth rate of *Mytilus edulis* in relation to water flow was investigated by Langan & Howell (1994) who found that the growth rate over 24 days was 0.1, 1.8, 2.0, 1.9 and 1.5mm at flow rates of 0, 0.01, 0.02, 0.04 and 0.08 m/s respectively. The only growth rate found to be significantly different was at zero flow. However, the pattern did follow that predicted by the "inhalant pumping speed" hypothesis that suggested maximal growth at water speeds of about 0.02 m/s and decreased growth rates at higher and lower speeds (Langan & Howell 1994). Higher current speed brings food to the bottom layers of the water column, and hence near to the mussels, at a higher rate (Frechette *et al*., 1989). Frechette *et al*., (1989) developed a model based on measurements in the St. Lawrence River estuary (Québec). The model suggested that *Mytilus edulis* consumption rate depends on the flow of water.

Widdows *et al*., (2002) found that there was no change in filtration rate of *Mytilus edulis* between 0.05 and 0.8 m/s. They noted that their finding contradicted earlier work that found a marked decline in filtration rates from 0.05 to 0.25 m/s (Newell, 1999; cited in Widdows *et al*., 2002) but suggested that the difference might be caused in differences in population studied, as the earlier work was based in the USA and their study used mussels from the Exe estuary in the UK. Widdows *et al*., (2002) also noted that above 0.8 m/s the filtration rate declined mainly because the mussels became detached from the substratum in the experimental flume tank. Widdows *et al*., (2002) noted that their results were consistent with field observations, as mussels show preferential settlement and growth in areas of high flow, such as the mouth of estuaries and at the base of power station cooling systems (Jenner *et al*., 1998). They also reported that Jenner *et al*.,(1998; cited in Widdows *et al*., 2002) observed that biofouling of cooling water systems by mussels was only reduced significantly when mean current speeds reached 1.8-2.2 m/s and was absent at >2.9 m/s.

Increased flow rate increases the risk of mussels being detached from the bed and transported elsewhere where their chance of survival will be significantly reduced due to the risk of predation and siltation (Dare, 1976). It is the strength of the byssal attachment that determines the mussel's ability to withstand increases in flow rate. Flow rate itself has been shown to influence the strength and number of byssus threads that are produced by *Mytilus edulis* and other *Mytilus* spp. with mussels in areas of higher flow rate demonstrating stronger attachment (Dolmer & Svane, 1994; Alfaro, 2006). Dolmer & Svane (1994) estimated the potential strength of attachment for *Mytilus edulis* in both still water and flows of 1.94 m/sec, by counting the number of established byssus threads and measuring the strength of attachment of individual detached byssus threads. It was found that in still water the strength of the attachment was 21% of the potential strength whilst at 19.4 cm/sec it was 81 % of the potential strength, suggesting that *Mytilus edulis* has the ability to adapt the strength of its attachment based on flow rate. The mussels were then able to withstand storm surges up to 16 m/s. Young (1985) demonstrated that byssus thread production and attachment increased with increasing water agitation. She observed the strengthening of byssal attachments by 25% within eight hours of a storm commencing and an ability to withstand surges up to 16 m/s. However, it was concluded that sudden surges may leave the mussels susceptible to being swept away (Young, 1985) as they need time to react to the increased velocity to increase the attachment strength. *Mytilus edulis* beds could, therefore, adapt to changes in water flow at the pressure benchmark.

Alfaro (2006) found that when a sudden increase in flow (to 0.13 m/s) was experienced by *Perna canalicuulus* (another mussel species) in areas of low flow rate they were more susceptible to detachment than those that had been exposed to a higher flow rate. It was also noted that the individuals kept at higher water flows (e.g. 10 cm/sec) produced more byssus threads. The increased energy used for byssus production in the high flow environments may reduce the energy that is available for other biological activities (Alfaro, 2006).

Water flow also affects the settlement behaviour of larvae. Alfaro (2005) observed that larvae settling in a low water flow environment are able to first settle and then detach and reattach displaying exploratory behaviour before finally settling and strengthening their byssus threads. However, larvae settling in high flow environments did not display this exploratory behaviour. Pernet *et al*., (2003) found that at high velocities, larvae of *Mytilus* spp. were not able to able to exercise much settlement preference. It was thought that when contact with suitable substratum is made the larvae probably secure a firm attachment. Movement of larvae from low shear velocities, where they use their foot to settle, to high shear velocities where they use their byssal thread to settle was observed by Dobretsov & Wahl (2008).

Moderate water movement is beneficial to seaweeds as it carries a supply of nutrients and gases to the plants and removes waste products. Propagule dispersal, fertilization, settlement, and recruitment are also influenced by water movement (Pearson & Brawley, 1996). Increased water flow can increase scour through increased sediment movement. Small life stages of macroalgae are likely to be affected by removal of new recruits from the substratum reducing successful recruitment (Devinny & Volse, 1978) (see ‘siltation’ pressures). A reduction in water flow can cause a thicker boundary layer resulting in lower absorption of nutrients and CO_2 by the macroalgae. Slower water movement can also cause oxygen deficiency directly impacting the fitness of algae (Olsenz, 2011). Higher water flow rates increase mechanical stress on macroalgae by increasing drag. This can result in individuals being torn off the substratum. Jonsson *et al*. (2006) found that flow speed of 7-8 m/s completely dislodged *Fucus vesiculosus* individuals larger than 10 cm. Smaller individuals are likely to better withstand increased water flow as they experience less drag. The risk of dislodgement is greater where algae are attached to pebbles instead of bedrock (Isaeus, 2004). As water velocity increases, algae can flex and reconfigure to reduce the size of the alga when aligned with the direction of flow, this minimises drag and hence the risk of dislodgement (Denny *et al*., 1998; Boller & Carrington, 2007). These characteristics allow these species to persist on shores that experience a range of flow speeds. For example, *Mastocarpus stellatus* occurs at sites in Maine, USA experiencing peak Autumn flow speeds as measured by current meters of 9.2 m/s and 5.8 m/s. The habitat structure created by canopies and turfs reduce the effects of water flows on individuals by slowing and disrupting flow (Boller & Carrington, 2006) although this effect will be reduced in this biotope where *Fucus serratus* and red algae occur as scattered plants. The coralline crusts characterizing this biotope are securely attached and as these are flat are subject to little or no drag.

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

Patella vulgata inhabits a range of tidal conditions and is therefore, likely to tolerate a change in water flow rate. The streamlined profile of limpet shells is of importance in increasing their tolerance of water movement, and this is undoubtedly one factor in determining the different shape of limpets at different exposures. With increasing exposure to wave action the shell develops into a low profile reducing the risk of being swept away. The strong muscular foot and a thin film of mucus between the foot and the rock enables *Patella vulgata* to grip very strongly to the substratum (Fretter & Graham, 1994). The ability of limpets to resist accelerating, as distinct from constant currents, may set a limit to the kind of habitat which they can occupy and limit the size to which they can grow.

Littorina littorea is found in areas with water flow rates from negligible to strong, although populations exposed to different levels of flow may have adapted to local conditions. Increases in water flow rates above 6 knots may cause snails in less protected locations (e.g. not in crevices etc.) to be continually displaced into unsuitable habitat so that feeding may become sub-optimal. Thus, populations of *Littorina littorea* are likely to reduce. Shell morphology within littorinids varies according to environmental conditions, in sheltered areas, where *Carcinus maenas* is more prevalent, shell apertures are small to inhibit predation. In exposed areas the foot surface is larger to allow greater attachment and the shell spire is lower to reduce drag (Raffaelli 1982, Crothers, 1992).

Sensitivity assessment. The biotope is characteristic of exposed to moderately wave exposed conditions where water movement from wave action will greatly exceed the strength of any possible tidal flow. Based on the available evidence the characterizing species and associated macroalgae and animals are able to adapt to high flow rates and the biotope is therefore considered to be 'Not sensitive' to an increase in water flow at the pressure benchmark. A decrease in water flow may have some effects on recruitment and growth of filter feeders including *Mytilus edulis*, but this is not considered to be lethal at the pressure benchmark and resistance is therefore assessed as 'High' and resilience as 'High' by default, so that the biotope is considered to be 'Not sensitive'. Changes in water flow at the pressure benchmark may result in increased or decreased sediment deposition, these are not considered to alter the character of the biotope which is silted (Connor *et al*., 2004) and therefore subject to some scouring but may alter species richness of the small invertebrates associated with the turf. *Patella vulgata* may however be sensitive to increased deposition (see siltation pressures).

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

Emergence regime is a key factor structuring intertidal biotopes. Changes in emergence can lead to; greater exposure to desiccation, temperature and salinity variation, reduced levels of time for filter feeding and nutrient uptake and photosynthesising opportunities for the characterizing species. Changes in emergence can also alter competitive interactions and trophic interactions such as grazing and predation. This biotope occurs in the mid shore (Connor *et al*., 2004). Typically above this biotope a similar *Mytilus edulis* and *Fucus vesiculosus* dominated biotope (MytFves) may be found. In the lower eulittoral zone below is usually the biotope Ldig.Ldig dominated by the kelp *Laminaria digitata* (Connor *et al*., 2004).

Mytilus edulis beds are found at a wide range of shore heights from the strandline down to the shallow sublittoral (Connor *et al*., 2004). Their upper limits are controlled by temperature and desiccation (Suchanek, 1978; Seed & Suchanek 1992; Holt *et al*., 1998) while the lower limits are set by predation, competition (Suchanek, 1978) and sand burial (Daly & Mathieson, 1977). Mussels found higher up the shore display slower growth rates (Buschbaum & Saier, 2001) due to the decrease in time during which they can feed and also a decrease in food availability. It has been estimated that the point of zero growth occurs at 55% emergence (Baird, 1966) although this figure will vary slightly depending on the conditions of the exposure of the shore (Baird, 1966; Holt *et al*., 1998). Increasing shore height does, however, increase the longevity of the mussels due to reduced predation pressure (Seed & Suchanek, 1992; Holt *et al*., 1998), resulting in a wider age class of mussels found on the upper shore. The lower limit of *Mytilus* beds is mainly set by predation from *Asterias rubens* and *Carcinus maenas* which may increase with a decrease in emergence potentially reducing the lower limit or reducing the number of size classes and age of the mussels at the lower range of the bed (Saier, 2002).

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

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

Experimental grazer removal has allowed algae including *Palmaria palmata*, *Ceramium* sp. and *Osmundea* (as *Laurenci*a) *pinnatifida* to grow higher on the shore (during winter and damp summers) than usual suggesting that grazing also limits the upper shore extent of this biotope (Hawkins & Hartnoll, 1985). *Palmaria palmata* grew more abundantly higher up the shore following the massive mortality of molluscan grazers after the Torrey Canyon oil spill (Hawkins & Hartnoll, 1983). *Palmaria palmata* also grew more abundantly higher up the shore following the massive mortality of molluscan grazers after the Torrey Canyon oil spill (Hawkins & Hartnoll, 1983). These observations and further grazer removal experiments by Boaventura *et al*., (2003), indicate that grazing, in combination with physiological tolerances, limits the upper shore extent of biotopes characterized by red algal turfs on moderately and more exposed shores, where grazing is greater than on sheltered shores (Hawkins & Hartnoll, 1983, Boaventura *et al*., 2003). These results concord with other studies that show grazing and emersion stress limit the height to which red algal turfs can extend (Underwood, 1980, Boaventura, 2000).

Occurrence of encrusting coralline algae seems to be critically determined by exposure to air and sunlight. Colonies survive in damp conditions under algal canopies or in pools but not on open rock where desiccation effects are important. Increased emergence leading to drying out of shallow pools would reduce habitat suitability for this group. Spore release by the crusting coralline *Lithophyllum incrustans* is triggered by small changes in salinity and temperature and therefore changes in emergence may alter patterns in reproduction and recruitment (see relevant pressures for further information). However, this species does occur both high and low in the intertidal (Edyvean & Ford, 1986) and presumably such impacts are limited

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

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

Sensitivity assessment. The key characterizing species *Mytilus edulis* is likely to tolerate some increases in emergence as typically a biotope dominated by this species and *Semibalanus balanoides* occurs above this biotope. Other species better able to tolerate desiccation are likely to competitively displace *Fucus serratus* following increased emergence. A significant, long-term, increase in emergence is therefore considered likely to lead to replacement of this biotope with one that is similar but more typical of the changed conditions with less red algae. Red algae are intolerant of high levels of desiccation but basal crusts may allow individuals to persist in conditions that are unfavourable to frond development until the emergence regime is re-established. A decrease in emergence is likely to also be tolerated by *Mytilus edulis* but *Fucus serratus* may be replaced by *Laminaria digitata* leading to biotope reclassification. As emergence is a key factor structuring the distribution of biotopes on the shore, resistance to a change in emergence (increase or decrease) is assessed as 'Low'. Recovery is assessed as 'High', (based on the impacted *Fucus serratus* and red algae, rather than the un-impacted *Mytilus edulis*) and sensitivity is therefore assessed as 'Low'.

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

This biotope is found in habitats that are judged to be moderately exposed (Connor *et al*., 2004).

Mytilus edulis are able to increase the strength of their attachment to the substratum in more turbulent conditions (Price, 1982; Young, 1985). Young (1985) demonstrated an increase in strength of the byssal attachment by 25 % within 8 hours of a storm commencing. When comparing mussels in areas of high flow rate and low flow rate those at a higher flow rate exhibit stronger attachments than those in the areas of lower flow (Dolmer & Svane, 1994; Alfaro, 2006). Dolmer & Svane (1994) found that in still water the strength of the attachment was 21 % of the potential strength whilst at 1.94 m/sec it was 81 % of the potential strength. The mussels were then able to withstand storm surges up to 16 m/s. Alfaro (2006) also noted that the individuals kept at higher water flows produce more byssal threads. The increased energy used for byssus production in the high flow environments may reduce the energy that is available for other biological activities (Alfaro 2006). Whilst this clearly demonstrates the ability of mussels to adapt to the various conditions to avoid dislodgement, the mussels are unlikely to adapt instantly and a sudden increase in flow is likely to result in dislodgement (Young, 1985).

Large scale destruction of mussel beds has been reported in many areas such as the Wash, Morecambe Bay and the Wadden Sea (Holt *et al*., 1998) and it appears that because of this high wave exposure and destruction, reefs found in wave exposed areas are likely to be more dynamic (Nehls & Thiel, 1993). Furthermore, increased wave exposure leads to a higher risk of damage from drift logs (or other flotsam), which once they have destroyed a patch of mussels leave the mussels around that patch at a higher risk of erosion (Seed & Suchanek, 1992). Mussels with high abundance of epizoic and epiphytic (e.g. barnacles and macroalgae) growing on mussels are also more susceptible to removal in areas of high exposure due to increased drag caused by these fouling organisms (Suchanek, 1985; Seed & Suchanek, 1992). However, mussel beds are prevalent in areas of high wave exposure suggesting a high resilience despite destruction.

Blue mussels display a high resistance to increases in water flow, but the oscillatory water movement that occurs on shores of higher wave exposure is likely to have a higher impact due to the 'to and fro' motion which is more likely to weaken the attachments. Westerbom & Jattu (2006) found that in subtidal mussel beds, mussel densities increased with increasing wave exposure. The highest biomass was found in areas of intermediate exposure, potentially due to the larger mussels being removed at high wave exposure levels. It was suggested that the lower densities found in more sheltered areas were due to low recruitment, early post-recruitment mortality, increased predation or stagnant settlement on rocks. Furthermore, it was also noted that high sedimentation, which is more prevalent in sheltered areas, as there is less energy for re-suspension, prevents colonisation and result in the death of small mussels that are living close to the sediment surface by smothering and the clogging up of their feeding apparatus (Westerbom & Jattu, 2006). Therefore, colonisation of new space in sheltered areas could be slow, particularly in areas where there is low availability of adult mussels.

The above evidence is variable as different studies have examined beds that differ in habitat, wave exposure, substratum and mussel density. However general trends can be seen. In rocky habitats, increased wave exposure allows mussel to dominate and form beds, especially where the rock surface has a low slope. Where the beds are patchy or damaged (from natural or human activities) they are more susceptible to further damage as a result of wave action or storms (Seed & Suchanek, 1992; Brosnan & Crumrine, 1994). Multi-layered mussel beds are less susceptible to damage, especially where only the surface layer is removed. It has been noted that the build-up of mussel mud (pseudofaeces) under the bed can reduce the attachment of the bed to the underlying substratum. But in areas of wave exposure, the flow of water through the bed will probably prevent the 'mussel mud' accumulating.

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

As water velocity increases red algae can flex and reconfigure to reduce the size of the alga when aligned with the direction of flow, this minimises drag and hence the risk of dislodgement (Boller & Carrington, 2007). These characteristics allow these species to persist on shores that experience a range of wave action levels. Colonies of *Lithophyllum incrustans* appear to thrive in conditions exposed to strong water movement. Irvine & Chamberlain (1994) observe that the species is best developed on wave exposed shores.

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

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

Chemical Pressures

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

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

Contamination at levels greater than the benchmark may impact this biotope. The effects of contaminants on *Mytilus* sp. were extensively reviewed by Widdows & Donkin, (1992) and Livingstone & Pipe (1992). Widdows & Donkin (1992) list tolerances of *Mytilus edulis* adults and larvae but note that lethal responses give a false impression of high tolerance, since the adults can close their valves and isolate themselves from the environment for days. They suggested that sublethal effects e.g. shell growth and 'scope for growth' (SFG), are more sensitive indicators of the effects of contaminants. Reported effects of heavy metals follow.

- Adult 15 day LC_{50} to 50µg/l Cu (Widdows & Donkin, 1992).
- Crompton (1997) reported that adult bivalve mortalities occurred after 4-14 day exposure to 0.1-1 µg/l Hg, 1-10 µg/l Cu and Cd, 10-100 µg/l Zn but 1-10 mg/l for Pb and Ni.
- Widdows *et al.* (1995) reported 'no observed effect thresholds' on feeding or SFG in *Mytilus edulis* tissues of 150 µg Cd/g dry wt, 25 µg Cu/ g dry wt, (lethal at 60 µg Cu/g dry wt), 12 µg Hg/g dry wt, 10 mg Pb/g dry wt, and 300 µg Zn/g dry wt. However, the tissue

concentration of heavy metals at the sites studied was not high enough to reduce SFG significantly.

- Mussels were reported to be missing from an wider area than other shore organisms on a Cumbrian shore in the vicinity of a phosphate rich effluent outfall contaminated by a number of heavy metals (Holt *et al*., 1998).
- Adults are ca >10 fold more intolerant than larvae to Cu, petroleum hydrocarbons and sewage sludge (Widdows & Donkin, 1992) (see larval sensitivity).

Overall, *Mytilus edulis* is probably relatively tolerant of heavy metal contamination. But the potential mortality indicated above suggest an intolerance of intermediate.
Recovery may occur rapidly through good annual recruitment but examination of patches in beds of *Mytilus* sp. revealed that they may take many years to recover (see additional information below), depending on shore height, competition and environmental conditions. Repeated loss and recruitment results in a patchy distribution of mussels on the shore (Seed & Suchanek, 1992). Therefore, a recoverability of 'high' has been reported.

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

Hydrocarbon & PAH contamination	Not Assessed (NA) Q: NR A: NR C: NR	Not assessed (NA) Q: NR A: NR C: NR	Not assessed (NA) Q: NR A: NR C: NR

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

Contamination at levels greater than the benchmark may impact this biotope. Widdows & Donkin (1992) list tolerances of *Mytilus edulis* adults and larvae but note that lethal responses give a false impression of high tolerance, since the adults can close their valves and isolate themselves from the environment for days. They suggested that sublethal effects e.g. shell growth and 'scope for growth' (SFG), are more sensitive indicators of the effects of contaminants.

- Widdows *et al.* (1995) demonstrated that toxic hydrocarbons, primarily poly-aromatic hydrocarbons, made a large contribution the decline in SFG observed along the North Sea coast. Hydrocarbons reduce clearance rate through 'non-specific narcosis'.

- Mussel populations in Sullom Voe experienced moderate hydrocarbon pollution and a reduced SFG but had sufficient capacity to grow, reproduce and maintain a viable population (Widdows *et al.*, 1987).
- Widdows *et al.* (1987) examined the response of *Mytilus edulis* to high oil (water accommodated fraction of diesel oil) (125 ± 28 µg/l) and low oil (28 ±7 µg/l) over a 8 month period, and subsequent recovery. They observed a marked reduction in SFG (due to reduced feeding rate and food absorption efficiency), and a correlation between the reduction in SFG and the hydrocarbon tissue burden (Widdows *et al.*, 1987; Widdows & Donkin, 1992; Widdows *et al.*, 1995). Mussels exposed to high oil conditions showed a negative SFG and weight loss. During recovery, 22 days after removal to 'clean' seawater the high oil mussels depurated (removed) hydrocarbons more rapidly than low oil mussels, and showed an increased clearance rate and growth rate associated with 'catch-up' growth. Both high and low oil mussels recovered completely within 55 days.
- Widdows *et al.* (1987) also reported that high and low oil contamination of the experimental basins resulted in 100% mortality amongst mussels kept in the basins from autumn 1982 to summer 1983 and from spring 1983 to summer 1984 respectively.
- A sunflower oil tanker spill off the Anglesey coast resulted in ingestion of oil droplets and subsequent mortalities after spawning (Mudge *et al.*, 1993; Holt *et al.*, 1998).
- Bokn *et al.*, (1993) demonstrated that *Mytilus edulis* was lost from mesocosm experiments continuously dosed with 30.1 to 129.4 µg/l of the water accommodated fraction of diesel, and was the most intolerant of the intertidal species studied.
- *Mytilus edulis* dominated jetty piles immediately adjacent to an oil refinery effluent in Milford Haven, suggesting a high tolerance of hydrocarbon contamination (K. Hiscock, pers. comm.).

Overall, hydrocarbon tissue burden results in decreased SFG and in some circumstances may result in mortalities, reduced abundance or extent of *Mytilus edulis*.

Following the *Torrey Canyon* oil spill in 1967, oil and detergent dispersants affected high shore specimens of *Corallina officinalis* more than low shore specimens. Plants in deep pools were afforded some initial protection, although probably later affected by contaminated runoff. In areas of heavy spraying, however, *Corallina officinalis* was killed (Smith 1968). Intolerance to hydrocarbon pollution has been assessed to be high, as key structural and important characterizing coralline algal species will be lost and the biotope not be recognized in their absence. Hydrocarbon contamination, at levels greater than the benchmark, e.g. from spills of fresh crude oil or petroleum products, may cause significant loss of *Ulva* spp. However, the species tends to recover very rapidly from oil pollution incidents. For instance, after the Torrey Canyon tanker oil in 1967, grazing species were killed, and a dense flush of ephemeral green algae (*Ulva*, *Blidingia*) appeared on the rocky shore within a few weeks and persisted for up to one year (Smith, 1968).

In areas of moderate oil deposit, up to about 1/2cm thick, on rocks after the *Torrey Canyon* oil spill, limpets had survived unscathed over a month after the event and feeding continued even though a coating of oil smothered their food source of algae and diatoms (Smith, 1968). Limpets can ingest thick oil and pass it through their gut. However, thick layers of oil smothering individuals will interfere with respiration and spoil normal food supplies for *Patella vulgata*. After the *Braer* oil spill, in common with many other oil spills, the major impact in the intertidal zone was on the population of limpets and other grazers. In West Angle Bay, where fresh oil from the *Sea Empress* tanker reached rocky shores within one day of the spill, limpet mortality was 90% (Glegg *et al.*, 1999). Thus *Patella vulgata* has higher intolerance to fresh oil which has a high component of volatile hydrocarbons remaining. A significant reduction in the density of juvenile limpets was also observed at all sites known to have been oiled by the *Sea Empress* spill (Moore, 1997). In longer term studies into the environmental effects of oil refinery effluent discharged into Littlewick Bay, Milford Haven, the number of limpets, usually found in substantial numbers on this type of shore, were considerably reduced in abundance on areas close to the discharge (Petpiroon & Dicks, 1982). In particular only large individuals were found close to the outfall point and juveniles were completely absent, suggesting that observed changes in abundance resulted from effluent effects on larval stages rather than upon adults directly.

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

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

Contamination at levels greater than the benchmark may impact this biotope. The effects of contaminants on *Mytilus* sp. were extensively reviewed by Widdows & Donkin, (1992) and Livingstone & Pipe (1992). Mussels are suspension feeders and, therefore, process large volumes of water together with suspended particulates and phytoplankton. Mussels absorb contaminants directly from the water, through their diet and via suspended particulate matter (Widdows & Donkin, 1992), the exact pathway being dependant on the nature of the contaminant.

- Widdows and Donkin (1992) reported 50% mortality from a tissue burden of 20 µg/g TBT.
- Exposure of *Mytilus edulis* to detergent (BP1002) in seawater resulted in 100% mortality at 10 ppm detergent, although all survived at 5 ppm detergent (Smith, 1968).
- Liu & Lee (1975) reported a LC_{50} of 250 µg/l of the herbicide trifluralin in *Mytilus galloprovincialis*
- *Mytilus edulis* has been reported to bioaccumulate the insecticide ivermecten, although no adverse effects were observed (Cole *et al*., 1999).
- Biphenyl (a dye carrier) reduced the feeding rate of *Mytilus edulis* by 50% at 0.3 mg/l (Donkin *et al*., 1989).
- PCBs accumulate in gonads, although tissue concentrations are significantly reduced after spawning, although this may affect the next generation (Hummel *et al*., 1989; Holt *et al*., 1995).
- Significant increases in the incidence of tumours (neoplasia) were reported in the US Mussel Watch programme in the presence of higher concentrations of combustion related poly-aromatic hydrocarbons,*cis*-chlordane pesticides and cadmium (Hillman, 1993; Holt *et al*., 1998).
- *Mytilus edulis* survived in a power station cooling water culvert, exposed to 0.1-0.2 mg/l hypochlorite, although their growth rates were reduced by about a third. Mussels were able to recover in hypochlorite free periods between chlorination dosing (Thompson *et al*., 1997). *Mytilus edulis* and *Mytilus galloprovincialis* were reported to suffer 100% mortality after 15-135 days continuous exposure to 0.2-1.0 mg/l hypochlorite (Khalanski & Borget, 1980; cited in Thompson *et al*., 1997).
- Holt *et al*. (1995) also report that mussels may be absent from areas of high boating activity, presumably due to TBT.

Widdows *et al*. (1995) compared 'scope for growth' (SFG) and chemical contaminants in tissues of mussels from 26 coastal and 9 offshore sites around the United Kingdom. They noted that polar organics (probably derived from phytoplankton) accounted for some reduction in SFG, while organo-chlorides showed a significant correlation with an unexplained component of the decline in SFG. However, TBT levels were only high enough to cause an effect (<10% reduction in SFG) at 8 study sites (Widdows *et al*., 1995). *Mytilus edulis* is probably relatively tolerant of contaminants. Widdows & Donkin (1992) list tolerances of *Mytilus edulis* adults and larvae (but note that lethal responses give a false impression of high tolerance, since the adults can close their valves and isolate themselves from the environment for days. They suggest that sublethal effects (shell growth and 'scope for growth') are more sensitive indicators of the effects of contaminants. Also, adults are ca. 4 times more sensitive than larvae to TBT (Widdows & Donkin, 1992, see larval sensitivity).

Cole *et al*. (1999) suggested that herbicides were (not surprisingly) very toxic to algae and macrophytes. Hoare & Hiscock (1974) noted that with the exception of *Phyllophora* species, all red algae including encrusting coralline forms, were excluded from the vicinity of an acidified halogenated effluent discharge in Amlwch Bay, Anglesey and that intertidal populations of *Corallina officinalis* occurred in significant amounts only 600m east of the effluent. Chamberlain (1996) observed that although *Lithophyllum incrustans* was quickly affected by oil during the *Sea Empress* spill, recovery occurred within about a year. The oil was found to have destroyed about one third of the thallus thickness but regeneration occurred from thallus filaments below the damaged area.

Following the *Torrey Canyon* oil spill in 1967, oil and detergent dispersants affected high shore specimens of *Corallina officinalis* more than low shore specimens. Plants in deep pools were afforded some initial protection, although probably later affected by contaminated runoff. In areas of heavy spraying, however, *Corallina officinalis* was killed. (Smith 1968). Limpets are extremely intolerant of aromatic solvent based dispersants used in oil spill clean-up. During the clean-up response to the *Torrey Canyon* oil spill nearly all the limpets were killed in areas close to dispersant spraying. Viscous oil will not be readily drawn in under the edge of the shell by ciliary currents in the mantle cavity, whereas detergent, alone or diluted in seawater, would creep in much more readily and be liable to kill the limpet (Smith, 1968). A concentration of 5ppm killed half the limpets tested in 24 hours (Southward & Southward, 1978; Hawkins & Southward, 1992). Acidified seawater affects the motility of *Patella vulgata*. At a pH of 5.5 motility was reduced whilst submerged but individuals recovered when returned to normal seawater. At a pH of 2.5 total inhibition of movement occurred and when returned to normal seawater half had died (Bonner *et al*., 1993). Reduced motility reduces time for foraging and may result in decreased survival of individuals. Acidified seawater can also change the shell composition which will lead to a decrease in its protective nature and hence survival (Bonner *et al*., 1993). Short periods (48 hours) are unlikely to have much effect on a population but long periods (1 year) may cause reduced grazing and an increase in algal growth. However, seawater is unlikely to reach pH 2.5 therefore intolerance to slight changes in pH will be low. Hoare & Hiscock (1974) reported that in Amlwch Bay *Patella vulgata* was excluded from sites within 100-150m of the discharge of acidified, halogenated effluent.

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

The periostracum of *Mytilus edulis* was reported to concentrate uranium (Widdows & Donkin, 1992). Mussels have also been reported to bioaccumulate ^{106}Ru, ^{95}Zr, ^{95}Nb, ^{137}Cs and ^{90}Sr (Cole *et al*., 1999). While the above data demonstrates that *Mytilus edulis* can accumulate radionucleides, little information concerning the effects of radionucleides on marine organisms was found. Algae bioaccumulate radionuclides (with extent depending on the radionuclide and the algae species). Adverse effects have not been reported at low levels and hence resistance is assessed as 'High' and resilience as 'High' (by default) so that the biotope is considered to be 'Not sensitive' at the pressure benchmark.Sensitivity to this pressure is therefore not assessed based on lack of evidence.

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

This pressure is **Not assessed**.

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

This biotope would only be exposed to low oxygen in the water column intermittently during periods of tidal immersion. In addition, in areas of wave exposure and/or moderately strong current flow, low oxygen levels in the water are unlikely to persist for very long as oxygen levels will be recharged by the incorporation of oxygen in the air into the water column or flushing with oxygenated waters.

Mytilus edulis is regarded as euryoxic, tolerant of a wide range of oxygen concentrations including zero (Zandee *et al*., 1986; Wang & Widdows 1991; Gosling, 1992; Zwaan de & Mathieu 1992; Diaz & Rosenberg 1995; Gray *et al*., 2002). Diaz & Rosenberg (1995) suggest it is resistant to severe hypoxia. Adult mytilids exhibited high tolerance of anoxia in laboratory tests, e.g. Theede *et al*., (1969) reported LD50of 35 days for *Mytilus edulis* exposed to 0.21 mg/l O2 at 10 °C, which was reduced to 25 days with the addition of sulphide (50 mg/l $Na_2S.9H_2O$). Jorgensen (1980) observed, by diving, the effects of hypoxia (0.2 -1 mg/l) on benthic macrofauna in marine areas in Sweden over a 3-4 week period. Mussels were observed to close their shell valves in response to hypoxia and survived for 1-2 weeks before dying (Cole *et al*., 1999; Jorgensen, 1980). All life stages show high levels of tolerance to low oxygen levels. *Mytilus edulis* larvae, for example, are tolerant down to 1.0 ml/l, and although the growth of late stage larvae is depressed in hypoxic condition, the settlement behaviour does not seem to be affected (Diaz & Rosenberg, 1995). Based on the available evidence *Mytilus edulis* are considered to be resistant to periods of hypoxia and anoxia although sub-lethal effects on feeding and growth may be expected.

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

Experiments have shown that thallus discs of *Ulva lactuca* plants can survive prolonged exposure to anoxia and hypoxia (Vermaat & Sand-Jensen, 1987; Corradi *et al*., 2006). Following resumption of normal oxygen conditions gametes were produced. The associated invertebrate species also show high tolerances for reduced oxygen at levels that exceed the pressure benchmark. *Littorina* *littorea* can easily survive 3-6 days of anoxia (Storey *et al*., 2013). *Semibalanus balanoides* can respire anaerobically, so they can tolerate some reduction in oxygen concentration (Newell, 1979). When placed in wet nitrogen, where oxygen stress is maximal and desiccation stress is low, *Semibalanus balanoides* have a mean survival time of 5 days (Barnes *et al*., 1963). Limpets can also survive for a short time in anoxic seawater; Grenon & Walker, (1981) found that in oxygen free water limpets could survive up to 36 hours, although Marshall & McQuaid (1989) found a lower tolerance for *Patella granularis*, which survived up to 11 hours in anoxic water. *Patella vulgata* and *Littorina littorea* are able to respire in air, mitigating the effects of this pressure during the tidal cycle.

Sensitivity assessment *Mytilus edulis* is considered to be 'Not Sensitive' to de-oxygenation at the pressure benchmark and the assessment is largely based on this species. As the biotope will only be exposed to this pressure when emersed and wave action will re-oxygenate waters while respiration will occur in air, biotope resistance was assessed as 'High' and resilience as 'High' (no effect to recover from), resulting in a sensitivity of 'Not sensitive'.

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

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

Nutrient enrichment may impact mussel beds by altering the biomass of phytoplankton and macroalgae. At low levels, nutrient enrichment may stimulate the growth of phytoplankton used as food - a potential beneficial effect. In the Wadden Sea, where fishing had caused the destruction of the local population of *Sabellaria spinulosa*, *Mytilus edulis* was able to colonize, partly because of the increase in coastal eutrophication (Maddock, 2008). Conversely, Dinesen *et al*. (2011) observed that a reduction in nutrient loading to comply with the WFD resulted in a decrease of mussel biomass in estuaries.

High levels of enrichment may stimulate algal blooms and macroalgal growth. The growth of macrophytes on the mussel beds may result in increased drag on the mussel bed and hence increase susceptibility to damage from wave action and/or storms (see changes in wave exposure pressure). Algal blooms may die off suddenly, causing de-oxygenation (see de-oxygenation pressure) where the algae decompose on the seabed. The thresholds at which these blooms occur depend on site-specific conditions and be mitigated by the degree of mixing and tidal exchange. Some algae have been shown to negatively affect *Mytilus edulis* when present in high concentrations. For example, blooms of the algae *Phaeocystis* sp., have been observed to block the mussels gills when present in high concentrations reducing clearing rates, and at high levels they caused a complete cessation of clearance (Smaal & Twisk, 1997). Blockage of the gills is also likely to reduce ingestion rates, prevent growth and cause reproductive failure (Holt *et al*., 1998). Other species known to negatively impact *Mytilus edulis* are *Gyrodinium aureolum* (Tangen, 1977; Widdows *et al*., 1979b) and non-flagellated chrysophycean alga (Tracey, 1988). The accumulation of toxins from algal blooms has also been linked to outbreaks of paralytic shellfish poisoning resulting in the closure of shell fish beds (Shumway, 1990).

Stimulation of annual ephemerals may accentuate the competition for light and space and hinder perennial species development or harm their recruitment (Kraufvelin *et al*., 2007). Krauflin *et al*. (2006) found only minor effect on the fucoid community structure as a response to high nutrient levels during the first 3 years of an enrichment experiment. During the 4th year of exposure however, *Fucus serratus* started to decline and population consequently crashed in the 5th year. The study observed full recovery of algal canopy and animal community in less than 2 year after conditions returned to normal. The results indicate that established rocky shore communities of perennial algae with associated fauna are able to persist for several years, even at very high nutrient levels, but that community shifts may suddenly occur if eutrophication continues. They also indicate that rocky shore communities have the ability to return rapidly to natural undisturbed conditions after the termination of nutrient enhancement.

The associated green algae species have been identified worldwide as species that occur in areas subject to increased nutrient input within the vicinity of sewage outfalls and at intermediately polluted sites (Bellgrove *et al*., 2010; Littler & Murray, 1975; May, 1985; Brown *et al*., 1990; Bellgrove *et al*., 1997). Grazers in the biotope may also benefit from increased availability of food resources, due to enhanced growth.

Atalah & Crowe (2010) added nutrients to rockpools occupied by a range of algae including encrusting corallines, turfs of *Mastocarpus stellatus*, *Chondrus crispus* and *Corallina officinalis* and green and red filamentous algae. The invertebrates present were mostly *Patella ulyssiponensis*, the winkle *Littorina littorea* and the flat top shell *Gibbula umbilicalis*. Nitrogen and phosphorous enhancement was via the addition of fertilisers, as either 40 g/litre or 20 g/litre. The treatments were applied for seven month and experimental conditions were maintained every two weeks. The experimental treatments do not directly relate to the pressure benchmark but indicate some general trends in sensitivity. The cover of green filamentous algae was significantly increased both by reduced grazing and increased nutrients, although the effect size was synergistically magnified by the combined effect of grazer removal and nutrients. Nutrient enrichment caused an absolute increase in the average cover of green filamentous algae of 19% (±3.9 S.E.) respect to the control treatments while the cover of red turfing algae was not affected by nutrient addition (Atalah & Crowe, 2010)

Sensitivity assessment. The pressure benchmark is relatively protective and may represent a reduced level of nutrient enrichment in previously polluted areas. *Mytilus edulis* beds are considered to be not sensitive to nutrient enrichment at levels that comply with the requirements for good status for transitional and coastal water bodies (UKTAG, 2014). Due to the tolerance of high levels of nutrient input demonstrated generally e.g. Bellgrove *et al*., (2010) and Atalah & Crowe, (2010), resistance to this pressure is assessed as 'High' and resilience as 'High' so that the biotope is assessed as 'Not sensitive'. Where *Corallina* dominated biotopes have replaced canopy forming species in enriched areas it is not clear whether a change to the benchmark would lead to a shift in biotope type. Once established the presence of *Corallina* spp. and other turf forming species may limit recruitment by taller species (Bellgrove *et al*., 2010). No evidence was found to support an assessment of this indirect effect and it is not presented within the table.

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

It should be noted that biotopes occurring in tide swept or wave exposed areas are less likely to experience the effects of organic enrichment as the organic matter will be rapidly removed. It has been demonstrated that regardless of the concentration of organic matter *Mytilus edulis* will maintain its feeding rate by compensating with changes to filtration rate, clearance rates, production of pseudofaeces and absorption efficiencies (Tracey, 1988; Bayne *et al*., 1993; Hawkins *et al*., 1996). A number of studies have highlighted the ability of *Mytilus edulis* to utilise the increased volume of organic material available at locations around salmon farms. Reid *et al*.,

(2010) noted that *Mytilus edulis* could absorb organic waste products from a salmon farm with great efficiency. Increased shell length, wet meat weight, and condition index were shown at locations within 200m from a farm in the Bay of Fundy allowing a reduced time to market (Lander *et al*., 2012). *Mytilus edulis* have also been recorded in areas around sewage outflows (Akaishi *et al*., 2007; Lindahl & Kollberg, 2008; Nenonen *et al*., 2008; Giltrap *et al*., 2013) suggesting that they are highly tolerant of the increase in organic material that would occur in these areas.

Organic enrichment and nutrient enrichment commonly co-occur, for example sewage deposits or outputs from fish farms may enhance nitrogen and phosphorous and organic matter. Nutrient enrichment alters the selective environment by favouring fast growing, ephemeral species such as *Ulva lactuca* and *Ulva intestinalis* (Berger *et al*., 2004, Kraufvelin, 2007). Rohde *et al*., (2008) found that both free growing filamentous algae and epiphytic microalgae can increase in abundance with nutrient enrichment. This stimulation of annual ephemerals may accentuate the competition for light and space and hinder perennial species development or harm their recruitment (Berger *et al*., 2003; Kraufvelin *et al*., 2007). 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). High nutrient levels may directly inhibit spore settlement and hinder the initial development of *Fucus vesiculosus* (Bergström *et al*., 2003). Bellgrove *et al*. (2010) found that coralline turfs out-competed fucoids at a site associated with organic enrichment caused by an ocean sewage outfall.

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

Sensitivity assessment. Based on resistance to sedimentation, exposure to wave action and observations of *Mytilus edulis* and turf forming algae thriving in areas of increased organic matter (Lander *et al*., 2012, Reid *et al*., 2010), it was considered that this biotope had 'High' resistance to increased organic matter at the pressure benchmark (which represents enrichment rather than gross pollution). Resilience is therefore assessed as 'High' (no effect to recover from) and the biotope is considered to be 'Not sensitive'.

Physical Pressures

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

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

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

This biotope is characterized by the hard rock substratum to which the characterizing *Mytilus edulis*, *Fucus serratus* and associated species such as red and green algae, barnacles limpets and anemones can firmly attach. A change to a sedimentary substratum would significantly alter the character of the biotope and would lead to the development of a biological assemblage more typical of the changed conditions. A change to an artificial substratum could also impact the development of this biotope as species may have settlement preferences for particular surface textures. *Mytilus edulis* can be found on a wide range of substrata including artificial substratum (e.g. metal, wood, concrete), bedrock, biogenic reef, caves, crevices / fissures, large to very large boulders, mixed, muddy gravel, muddy sand, rock pools, sandy mud, small boulders, under boulders (Connor *et al*., 2004). An increase in the availability of hard substratum may be beneficial in areas where sedimentary habitats were previously unsuitable for colonisation e.g. coarse, mobile sediments. However, artificial hard substratum may also differ in other characteristics from natural hard substratum, so that replacement of natural surfaces with artificial may lead to changes in the biotope through changes in species composition, richness and diversity (Green *et al*., 2012; Firth *et al*., 2014) or the presence of non-native species (Bulleri & Airoldi, 2005).

Experiments indicate that there are species-specific preferences for substratum type among algae. Tests with stone panels fixed to the sublittoral, mid-tide and high-tide levels of varying roughness found that *Ulva* species settle preferentially on smoother, fine grained substratum (chalk, mottled sandstone) and *Porphyra purpurea* on rougher, granulated substratum (limestone, granite, basaltic larvae) (Luther, 1976).

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

Sensitivity assessment. The sensitivity assessment is based on a change to a soft sedimentary habitat. As this would remove the habitat for this biotope, resistance is assessed as ‘None’ and resilience is judged to be ‘Very Low’, as the change is considered to be permanent. Sensitivity is therefore assessed as 'High'.

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

Not relevant to biotopes occurring on bedrock.

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

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

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

The species characterizing this biotope occur on the rock and therefore have no shelter from abrasion at the surface. Most macroalgae are very flexible but not physically robust. Fucoid algae are particularly intolerant of trampling, depending on intensity. Fucoid algae demonstrate a rapid (days to months) detrimental response to the effects of trampling, depending on species, which has been attributed to either the breakage of their fronds across rock surfaces (Boalch *et al*., 1974) or their possession of small discoid holdfasts that offer little resistance to repeated impacts (Brosnan & Crumrine, 1992; Fletcher & Frid, 1996b). Foliose species such as *Mastocarpus* spp. are also likely to be intolerant of trampling (Brosnan & Crumrine, 1994). Brosnan (1993) suggested that the presence or absence of foliose algae (e.g. fucoids) could be used to indicate the level of trampling on the rocky shores of Oregon.

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

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

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

Brosnan & Crumrine (1994) noted that trampling significantly reduced algal cover within 1 month of trampling. Foliose algae were particularly affected and decreased in cover from 75% to 9.1% in trampled plots. *Mastocarpus papillatus* decreased in abundance from 9% to 1% in trampled plots but increased in control plots. *Fucus distichus* decreased in the summer months only to recover in winter but in trampled plots remained in low abundance (between 1 and 3% cover). Trampling resulted in a decrease in the cover of *Pelvetiopsis limitata* from 16% to 1.5%. *Iridaea cornucopiae* decreased from 38 to 14% cover within a month and continued to decline to 4-8% cover. However, after trampling ceased, recovery of algal cover including *Iridaea cornucopiae* and *Mastocarpus papillatus* was rapid (ca 12 months) (Brosnan & Crumrine, 1994). Schiel & Taylor (1999) also observed a decrease in understorey algae (erect and encrusting corallines) after 25 or more tramples, probably due to an indirect effect of increased desiccation as above. However, Schiel & Taylor (1999) did not detect any variation in other algal species due to trampling effects. Similarly, Keough & Quinn (1998) did not detect any effect of trampling on algal turf species. In general, studies show that turf-forming algae appear to be relatively resistant to single events and low levels of trampling. Brosnan & Crumrine (1994), for example, found that in experimentally trampled plots the cover of foliose and canopy forming species declined while turf-forming algae were relatively resistant.

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

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

Mytilus edulis lives on the surface of the seabed held by byssus threads attached to either the substratum or to other mussels in the bed. Activities resulting in abrasion and disturbance can either directly affect the mussel by crushing them or indirectly affect them by the weakening or breaking of their byssus threads making them vulnerable to displacement (Denny, 1987) where they are unlikely to survive (Dare, 1976). In addition, abrasion and sub-surface damage may attract mobile scavengers and predators including fish, crabs, and starfish to feed on exposed, dead and damaged individuals and discards (Kaiser & Spencer, 1994; Ramsay *et al*., 1998; Groenewold & Fonds, 2000; Bergmann *et al*., 2002). This effect will increase predation pressure on surviving damaged and intact *Mytilus edulis*. A number of activities or events that result in abrasion and disturbance and their impacts on mussel beds are described below, based on the review by Mainwaring *et al*. (2014).

In general, studies have found that trampling is an additional disturbance to the natural disturbances that the intertidal organisms are adapted to tolerate. Large declines of *Mytilus californianus* from mussel beds due to trampling have been reported (Brosnan, 1993; Brosnan & Crumrine, 1994; Smith & Murray, 2005). Brosnan & Crumrine (1994) recorded the loss of 54% of mussels from a single experimental plot on one day. Mussels continued to be lost throughout the experimental period, forming empty patches larger than the experimental plots. The empty patches continued to expand after trampling had ceased, due to wave action. At another site, the mussel bed was composed of two layers, so that while mussels were lost, cover remained. Brosnan (1993) also reported a 40 % loss of mussels from mussel beds after three months of trampling and a 50 % loss within a year. Van de Werfhorst & Pearse (2007) examined *Mytilus californianus* abundance at sites with differing levels of trampling disturbance. The highest percentage of mussel cover was found at the undisturbed site while the severely disturbed site showed low mussel cover. Brosnan and Crumrine (1994) noted that mussels that occupied hard substrata but did not form beds were also adversely affected. Although only at low abundance (2.5 % cover), all mussels were removed by trampling within 4 months. Brosnan & Crumrine (1994) noted that mussels were not common and confined to crevices in heavily trampled sites. Similarly, the mussel bed infauna (e.g. barnacles) was adversely affected and were crushed or lost with the mussels to which they were attached. However, Beauchamp & Gowing (1982) did not observe any differences in mussel density between sites that differed in visitor use.

Smith & Murray (2005) examined the effects of low-level disturbance on an extensive bed of *Mytilus californianus* (composed of a single layer of mussels) in southern California. Smith & Murray (2005) reported that in experimental plots exposed to trampling, mussel loss was 20-40 % greater than in untreated plots. A decrease in mussel mass, density, cover and maximum shell length were recorded even in low intensity trampling events (429 steps/m2). However, only 15 % of mussel loss was as a direct result of trampling, with the remaining loss occurring during intervals between treatment applications. Brosnan & Crumrine (1994) suggested that trampling destabilizes the mussel bed, making it more susceptible to wave action, especially in winter. Smith & Murray (2005) suggested that an indirect effect of trampling was weakening of byssal threads, which increases mussel susceptibility to wave disturbance (Denny, 1987). Brosnan & Crumrine (1994) observed recruitment within experimental plots did not occur until after trampling had ceased, and no recovery had occurred within 2 years

Paine & Levine (1981) examined natural patch dynamics in a *Mytilus californianus* bed in the USA. They suggested that it may take up to seven years for large barren patches to recover. However, chronic trampling may prevent recovery altogether. This would result in a shift from a mussel dominated habitat to one dominated by an algal turf or crust (Brosnan & Cumrine, 1994), completely changing the biotope. However, a small period of trampling could allow communities to recover at a similar rate to that of natural disturbance as the effects are similar. The associated epifauna and epiflora suffer the greatest amount of damage as they are the first organisms that a foot makes contact with (Brosnan & Crumrine, 1994). The loss of epifauna and epiflora could initially be of benefit to the mussel bed, despite the obvious decrease in species diversity, as there will be a decrease in drag for the mussels reducing the risk of dislodgement (Witman & Suchanek 1984) and freeing up more energy for growth and reproduction. However, it is likely that after continued trampling this effect will be minimal compared with the increased risk of dislodgement caused by trampling. No studies assessing the effect of trampling on mussels on intertidal muddy sand or sediments were found. Losses to the adult mussels by crushing or by suffocation where these are forced into the sediment are expected. There is the potential that this will open up areas for new recruitment or it may just create a similar situation to that seen on the rocky shore where wave damage and continual trampling prevent settlement and recovery.

The collision of objects, such as wave driven logs (or similar flotsam), is known to cause the removal of patches of mussels from mussel beds (Seed & Suchanek, 1992; Holt *et al*., 1998). When patches occur in mussel beds a good recruitment could result in a rapid recovery or the patch may increase in size through the weakening of the byssus threads of the remaining mussels leaving them vulnerable to erosion from storm damage (Denny, 1987).

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

The effects of trampling on barnacles appear to be variable with some studies not detecting significant differences between trampled and controlled areas (Tyler-Walters & Arnold, 2008). However, this variability may be related to differences in trampling intensities and abundance of populations studied. The worst case incidence was reported by Brosnan & Crumrine (1994) who found that a trampling pressure of 250 steps in a 20x20 cm plot one day a month for a period of a year significantly reduced barnacle cover (*Semibalanus glandula* and *Chthamalus dalli*) at two study sites. Barnacle cover reduced from 6 6% to 7 % cover in 4 months at one site and from 21 % to 5 % within 6 months at the second site. Overall barnacles were crushed and removed by trampling. Barnacle cover remained low until recruitment the following spring. Long *et al*. (2011) also found that heavy trampling (70 humans /km/hrs) led to reductions in barnacle cover. Single step experiments provide a clearer, quantitative indication of sensitivity to single events of direct abrasion. Povey & Keough (1991) in experiments on shores in Mornington peninsula, Victoria, Australia, found that in single step experiments 10 out of 67 barnacles, (*Chthamlus antennatus* about 3mm long), were crushed. However, on the same shore, the authors found that limpets may be relatively more resistant to abrasion from trampling. Following step and kicking experiments, few individuals of the limpet C*ellana trasomerica*, (similar size to *Patella vulgata*) suffered damage or relocated (Povey & Keough, 1991). One kicked limpet (out of 80) was broken and 2 (out of 80) limpets that were stepped on could not be relocated the following day (Povey & Keough, 1991). On the same shore, less than 5% of littorinids were crushed in single step experiments (Povey & Keough, 1991).

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

Sensitivity assessment. The available evidence indicates that abrasion could cause a significant loss of *Mytilus edulis* and fucoid cover and a reduction in species abundance and diversity. Based on the available evidence it is concluded that the biotope is sensitive to abrasion and that resistance of characterizing and associated species is 'Low' (loss of 25-75% individuals within direct impact footprint), resilience is assessed as 'Medium' (based on *Mytilus edulis*), resulting in a sensitivity of 'Medium'.

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

The species characterizing this biotope group are epifauna or epiflora occurring on rock, which is resistant to subsurface penetration. Therefore, 'penetration' is **'Not relevant'**. The assessment for abrasion at the surface only is, therefore, considered to equally represent sensitivity to this pressure'. Please refer to 'abrasion' above.

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

Intertidal biotopes will only be exposed to this pressure when submerged during the tidal cycle, and will thus experience limited exposure. Siltation, which may be associated with increased suspended solids and the subsequent deposition of these is assessed separately (see siltation pressures). In general, increased suspended particles reduce light penetration and increase scour and deposition. Changes in suspended solids may enhance food supply to filter or deposit feeders (where the particles are organic in origin), or decrease feeding efficiency (where the particles are inorganic and require greater filtration efforts). Increased suspended solid supply in sheltered habitats may lead to deposition on the fronds of *Fucus serratus* and other macroalgae, which could potentially reduce photosynthesis and growth rate. Although this biotope occurs in areas sheltered from wave action, it may experience strong tidal streams: water action could, therefore, remove deposited sediments. Although this removal may lead to abrasion, it will also reduce the level of siltation preventing effects on photosynthesis and reducing settlement on rock and other surfaces.

Changes in suspended solids affecting water clarity can have a direct impact on photosynthesis in *Fucus serratus*. Irradiance below the light compensation point of photosynthetic species can compromise growth (Middelboe *et al*., 2006). However, turbidity is only relevant when the biotope is covered with water as seaweed photosynthesis declines on emersion and recommences when recovered with water.

Sediment deposition can also interfere with attachment of microscopic stages of seaweeds reducing recruitment (see 'siltation' pressures). In extreme turbidity, such as found in the Bristol Channel, *Fucus serratus* is excluded from the bottom of the intertidal (below 2 m above chart datum) due to the lack of light for sustained growth (Chapman, 1995). On sites affected by high levels of resuspended colliery waste particles in the north-east of England, Hyslop *et al*., (1997) found that *Palmaria palmata* and *Ulva* spp. were reduced or absent, although the more tough fucoids were less affected. It is not clear how the levels of suspended solids experienced by these sites relate to the pressure benchmark. Increases in the cover of sediment trapping, turf-forming red algae at the expense of canopy forming species have been observed worldwide in temperate systems and have been linked to increased suspended solids linked to human activities worldwide (Airoldi, 2003). Increased suspended sediment may also result in increased scour, which may adversely affect *fucoids* and foliose red algae, and interfere with settling spores and recruitment if the factor is coincident with their major reproductive period. However, coralline algae, especially the crustose forms are thought to be resistant of sediment scour (Littler & Kauker, 1984), and will probably not be adversely affected at the benchmark level.

The biotope occurs in shallow waters where light attenuation due to increases in turbidity is probably low when submerged. Red algae and coralline algae especially are known to be shade tolerant and are common components of the understorey on seaweed dominated shores. Limited shading from suspended sediments is therefore not considered to negatively affect this genus. Other red algal species have high tolerances for high levels of suspended solids. *Chondrus crispus* occurs in areas of sand covered rock in the subtidal biotope IR.HIR.KSed.ProtAhn, suggesting it is very resistant to high levels of turbidity and scour associated with high levels of resuspended particles.

Mytilus edulis does not rely on light penetration for photosynthesis. In addition, visual perception is limited and the species does not rely on sight to locate food or other resources. An indirect effect of increased turbidity and reduced light penetration may be reduced phytoplankton productivity which could reduce the food availability for *Mytilus edulis*. However, as *Mytilus edulis* uses a variety of food sources and food is brought in from other areas with currents and tides, the effect is likely to be minimal. This species and the biotopes it forms are therefore not sensitive to changes in water clarity that refer to light penetration. Some siltation may occur within this biotope (Connor *et al*., 2004) suggesting that the biotope may experience high levels of suspended sediment from resuspension.

Mytilus edulis are often found in areas with high levels of turbidity. For example, the average suspended particulate matter (SPM) concentration at Hastings Shingle Bank was 15 -20 mg/l in June 2005, reaching 50 mg/l in windier (force 4) conditions, although a concentration of 200 mg/l was recorded at this site during gales (Last *et al*., 2011).

Winter (1972, cited by Moore, 1977) recorded 75 % mortality of *Mytilus edulis* in concentrations of 1.84-7.36 mg/l when food was also available. However, a relatively small increase in SPM concentration e.g. from 10 mg/l to 90 mg/l was found to increase growth rates (Hawkins *et al*., 1996). Concentrations above 250 mg/l have been shown to impair the growth of filter-feeding organisms (Essink, 1999). But Purchon (1937) found that concentrations of particulates as high a 440 mg/l did not affect *Mytilus edulis* and that mortality only occurred when mud was added to the experiment bringing the concentrations up to 1220 mg/l. The reason for some of the discrepancy between studies may be due to the volume of water used in the experiment. Loosanoff (1962) found that in small quantities of turbid water (due to particulates) the mussel can filter out all of the particulates within a few minutes whereas in volumes >50 gallons per individual the mussel becomes exhausted before the turbidity has been significantly lowered, causing it to close its shell and die.

It may be possible for *Mytilus edulis* to adapt to a permanent increase in SPM by decreasing their gill size and increasing their palp size in areas of high turbidity (Theisen, 1982; Essink, 1999). In areas of variable SPM, it is likely that the gill size would remain the same but the palp would adapt (Essink, 1999). Whilst the ability to adapt may prevent immediate declines in health, the energetic costs of these adaptations may result in reduced fitness; the extent of which is still to be established.

Mytilus edulis uses the circadian clock to determine the opening of the shell gape in nocturnal gape cycles (Ameyaw-Akumfi & Naylor, 1987). Last *et al*., (2011) investigated the effects on increased SPM concentrations on both the gape pattern and mortality in order to establish the effect that aggregate dredging will have on *Mytilus edulis* and other benthic invertebrates. Therefore they tested concentrations similar to those expected within a few hundred meters of an aggregate extraction site. The highest concentration tested using a pVORT (paddle VOrtex Resuspension Tanks) was ~71 mg/l. They showed that there is a significant reduction of the strength of the nocturnal gape cycle at high suspended sediment loads as well as a change in the gape period. The effects of these changes are not fully known but as it is likely that the gape pattern is a strategy to avoid diurnal predators the change may result in an increased risk of predation. After continued measurements of the gape cycle for 4 days post treatment, Last *et al*., (2011) observed that the cycle took longer than this to recover from the cycle disruption. Further study is required to determine the length of time required for recovery of this behavioural response (Last *et al*., 2011).

Based on a comprehensive literature review, Moore (1977) concluded that *Mytilus edulis* displayed a higher tolerance to high SPM concentrations than many other bivalves although the upper limit of this tolerance was not certain. He also hypothesised that the ability of the mussel to clean its shell in such conditions played a vital role in its success along with its pseudofaecal expulsion.

Sensitivity assessment. The exposure of this biotope to suspended sediments in the water column will be limited to immersion periods, and wave action will reduce accumulation. The biotope is considered to be 'Not sensitive' (resistance and resilience are 'High') to a reduction in suspended solids, although this may reduce food supply to *Mytilus edulis*, barnacles and other filter and deposit feeders that occur in this biotope. The available evidence suggests that *Mytilus edulis* is likely to tolerate an increase in suspended sediment at the pressure benchmark (an increase from clear to intermediate on the UK TAG, 2014 scale). An increase in suspended solids may lead to some sub-lethal abrasion of fronds of *Fucus serratus* and some reduction in photosynthesis while submerged with some effects on recruitment. Evidence globally indicates that increase suspended solids favour the turf-forming algae that occur within this biotope (Airoldi, 2003). Resistance is therefore assessed as 'Low-Medium' and resilience as 'High' so that sensitivity of the biotope is considered to be 'Low'. A long-term increase in suspended solids may result in species replacements with an increase in species seen in variable or reduced salinity, very turbid, silty environments such as *Polyides rotunda*, *Ulva* spp and *Furcellaria lumbricalis*.

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

Sedimentation can directly affect assemblages inhabiting rocky shores through burial/smothering and scour or abrasion of organisms. This biotope occurs in areas sheltered from wave action but where tidal currents may be strong. In areas with greater water flow, excess sediments are likely to be removed from the rock surface within a few tidal cycles, reducing the time of exposure to this pressure.

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

Increased abundance of algal turfs worldwide has been linked to sediment perturbations although not all the pathways and mechanisms of these effects are clear (see review by Airoldi, 2003). However, even the most tolerant of organisms would eventually suffer from inhibition and mortality following smothering although the thresholds for these effects have has not been identified (Airoldi, 2003). In a review of the effects of sedimentation on rocky coast assemblages, Airoldi (2003) outlined the evidence for the sensitivity of coralline algae to sedimentation. The reported results are contradictory with some authors suggesting that coralline algae are negatively affected by sediments while others report that encrusting corallines are often abundant or even dominant in a variety of sediment impacted habitats (Airoldi, 2003 and references therein). Crustose corallines have been reported to survive under a turf of filamentous algae and sediment for 58 days (the duration of the experiment) in the Galapagos (species not identified, Kendrick, 1991). The crustose coralline *Hydrolithon reinboldii*, has also been reported to survive deposition of silty sediments on subtidal reefs off Hawaii (Littler, 1973).

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

Mytilus edulis occur in areas of high suspended particulate matter (SPM) and therefore a level of siltation is expected from the settling of SPM. In addition, the high rate of faecal and pseudofaecal matter production by the mussels naturally results in siltation of the seabed, often resulting in the formation of large mounds beneath the mussel bed. For example, at Morecambe Bay, an accumulation of mussel mud (faeces, pseudofaeces and washed sand) of 0.4-0.5m between May 1968 and September 1971 resulted in the mortality of young mussels (Daly & Mathieson, 1977). In order to survive the mussels needed to keep moving upwards to stay on the surface. Many individuals did not make it to the surface and were smothered by the accumulation of mussel mud (Daly & Mathieson, 1977) so that whilst *Mytilus edulis* does have the capacity to vertically migrate through sediment some individuals will not survive.

Sand burial has been shown to determine the lower limit of *Mytilus edulis* beds (Daly & Mathieson, 1977). Burial of *Mytilus edulis* beds by large-scale movements of sand and resultant mortalities have been reported from Morecambe Bay, the Cumbrian coast and Solway Firth (Holt *et al*., 1998). Essink (1999) recorded fatal burial depths of 1-2 cm for *Mytilus edulis* and suggested that they had a low tolerance of sedimentation based on investigations by R.Bijkerk (cited by Essink, 1999). Essink (1999) suggested that deposition of sediment (mud or sand) on shallow mussel beds should be avoided. However, Widdows *et al*. (2002) noted that mussels buried by 6 cm of sandy sediment (caused by resuspension of sediment due to turbulent flow across the bed) were able to move to the surface within one day. Conversely, Condie (2009) (cited by Last *et al*., 2011) reported that *Mytilus edulis* was tolerant of repeated burial events.

Last *et al*., (2011) carried out burial experiments on *Mytilus edulis* in pVORTs. They used a range of burial depths and sediment fractions and temperatures. It was found that individual mussels were able to survive burial in depths of 2, 5 and 7 cm for over 32 days although the deeper and longer the mussels were buried the higher the mortality. Only 16 % of buried mussels died after 16 days compared to almost 50 % mortality at 32 days. Mortality also increased sharply with a decrease in particle size and with increases in temperature from 8.0 and 14.5 to 20 °C. The ability of a proportion of individuals to emerge from burial was again demonstrated with approximately one quarter of the individuals buried at 2 cm resurfacing. However, at depths of 5 cm and 7 cm no emergence was recorded (Last *et al*., 2011). The lower mortality when buried in coarse sands may be related to the greater number of individuals who were able to emerge in these conditions and emergence was to be significant for survival.

It is unclear whether the same results would be recorded when mussels are joined by byssal threads or whether this would have an impact on survival (Last *et al*., 2011), although Daly & Mathieson (1977) recorded loose attachments between juvenile mussels during a burial event and some of these were able to surface. It was not clear whether the same ability would be shown by adult mussels in a more densely packed bed.

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

Sensitivity assessment. Burial will lower survival and germination rates of spores and cause some mortality in early life stages of *Fucus serratus* and foliose red algae. Adults are more resistant but will experience a decrease in growth and photosynthetic rates. Mortality will be more limited and possibly avoided, where the smothering sediment is removed due to wave action or tidal streams, depending on how long the sediment remains. Deposition of 5 cm of fine material (see benchmark) in a single incident is unlikely to result in significant mortality in blue mussel clumps before sediments are removed by current and wave action. However, the inability of *Mytilus edulis* to emerge from sediment deeper than 2 cm (Last *et al.*, 2011, Essink, 1999, Daly & Matthieson, 1977) and the increased mussel mortality with depth and reduced particle size observed by Last *et al.* (2011), suggest that there may be some mortality and resistance is assessed as 'Medium'. Survival will be higher in winter months when temperatures are lower and physiological demands are decreased. Resistance and resilience have both been assessed as 'Medium' based on *Mytilus edulis*. Overall the biotope has a 'Medium' sensitivity to smothering at the level of the benchmark. It should be noted that the associated *Patella vulgata* and littorinids may have higher sensitivities to this pressure.

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

Sensitivity to this pressure will be mediated by site-specific hydrodynamic conditions and the footprint of the impact. Where a large area is covered sediments may be shifted by wave and tides rather than removed. However, mortality will depend on the duration of smothering, where wave action rapidly mobilises and removes fine sediments, survival may be much greater. No evidence was found to assess this pressure at the benchmark. 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).

Mytilus edulis may be able to survive if the deposit was rapidly removed and the depth decreased. Last *et al.* (2011) carried out burial experiments on *Mytilus edulis* in pVORTs. They used a range of burial depths and sediment fractions and temperatures. It was found that individual mussels were able to survive burial in depths of 2, 5 and 7 cm for over 32 days although the deeper and longer the mussels were buried the higher the mortality. Only 16 % of buried mussels died after 16 days compared to almost 50 % mortality at 32 days. Mortality also increased sharply with a decrease in particle size and with increases in temperature from 8.0 and 14.5 to 20 °C. Several studies found that increasing the vertical sediment burden negatively impact fucoids survival and associated communities. Even the most tolerant of organisms would eventually suffer from inhibition and mortality following smothering although the thresholds for these effects have has not been identified (Airoldi, 2003).

Sensitivity assessment. At the level of the benchmark (30 cm of fine material added to the seabed in a single event) smothering is likely to result in mortalities of *Mytilus edulis* understorey algae, invertebrate grazers and young (germling) fucoids. Resistance is assessed as 'Low' as all individuals exposed to siltation at the benchmark level are predicted to die and a 'Medium' resilience. Overall the biotope has a 'Medium' sensitivity to siltation at the pressure benchmark.

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

Not assessed. *Mytilus edulis* ingest microplastics. A laboratory experiment using microbeads of polystyrene, demonstrated uptake of particles by *Mytilus edulis* within 12 hours (Browne *et al.*, 2008). After three days some of the beads were translocated to the circulatory system.

Microplastics were excreted in faecal pellets but were still present in hemolymph 48 days later. No toxicological effects were observed and there were no changes in filter feeding activity (Browne *et al*., 2008). As exposure was short-term it is not clear whether lethal or sub-lethal effects would occur in wild populations over extended periods. There is currently no evidence to assess the level of impact.

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

No evidence.

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

Not relevant.

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

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

Red algae, in general, are shade tolerant, often occurring under a macroalgal canopy that reduces light penetration. In areas of higher light levels, the fronds may be lighter in colour due to bleaching (Colhart & Johansen, 1973). Other red algae in the biotope are flexible with regard to light levels. Canopy removal experiments in a rocky subtidal habitat in Nova Scotia, Canada by Schmidt & Scheibling (2007) did not find a shift in understorey macroalgal turfs (dominated by *Corallina officinalis*, *Chondrus crispus* and *Mastocarpus stellatus*) to more light-adapted species over 18 months.

Coralline crusts 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. Samples of *Lithophyllum impressum* suspended from a raft and shaded (50-75% light reduction) continued to grow over two years (Dethier, 1994). In areas of higher light levels, the fronds and bases may be lighter in colour due to bleaching (Colhart & Johansen, 1973). Other red algae in the biotope are flexible with regard to light levels and can also acclimate to different light levels.

Sensitivity assessment. The loss of *Fucus serratus* would lead to biotope reclassification, therefore a biotope resistance of ' Low' is suggested albeit at low confidence. Resilience is likely to be 'High' so that sensitivity is 'Low.

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

No direct evidence was found to assess this pressure. As the larvae of *Mytilus edulis* are planktonic and are transported by water movements, barriers that reduce the degree of tidal excursion may alter the supply of *Mytilus edulis* to suitable habitats from source populations. However, the presence of barriers may enhance local population supply by preventing the loss of larvae from enclosed habitats. This species is therefore potentially sensitive to barriers that restrict water movements, whether this will lead to beneficial or negative effects will depend on whether enclosed populations are sources of larvae or are ‘sink’ populations that depend on outside supply of larvae to sustain the local population.The associated macroalgae (with the exception of *Ulva* spp.) have limited dispersal Barriers and changes in tidal excursion are not considered relevant to these species as dispersal is limited. As the key characterizing species *Mytilus edulis* are widely distributed and have larvae capable of long distance transport, resistance to this pressure is assessed as 'High' and resilience as 'High' by default. This biotope is therefore considered to be 'Not sensitive'.

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

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

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

Not relevant.

Biological Pressures

	Resistance	Resilience	Sensitivity
Genetic modification & translocation of indigenous species	No evidence (NEv) Q: NR A: NR C: NR	No evidence (NEv) Q: NR A: NR C: NR	No evidence (NEv) Q: NR A: NR C: NR

Two species of *Mytilus* occur in the UK, *Mytilus edulis* and *Mytilus galloprovincialis*. *Mytilus edulis* appears to maintain genetic homogeneity throughout its range whereas *Mytilus galloprovincialis* can be genetically subdivided into a Mediterranean group and an Atlantic group (Beaumont *et al.* 2007). *Mytilus edulis* and *Mytilus galloprovincialis* have the ability to hybridise in areas where their distribution overlaps e.g. around the Atlantic and European coast (Gardner, 1996; Daguin *et al.*, 2001; Bierne *et al.*, 2002; Beaumont *et al.*, 2004). In the UK overlaps occur on the North East coast, North East Scotland, South West England and in the North, West and South of Ireland (Beaumont *et al.*, 2007). It is difficult to distinguish *Mytilus edulis*, *Mytilus galloprovincialis* or hybrids based on shell shape because of the extreme plasticity of shape exhibited by mussels under environmental variation, and a genetic test is required (Beaumont *et al.*, 2007). There is some discussion questioning the distinction between the two species as the hybrids are fertile (Beaumont *et al*., 2007). Hybrids reproduce and spawn at a similar time to both *Mytilus edulis* and *Mytilus galloprovincialis* which supports genetic flow between the taxa (Doherty *et al*., 2009).

There is some evidence that hybrid larvae have a faster growth rate to metamorphosis than pure individuals which may leave pure individuals more vulnerable to predation (Beaumont *et al*., 1993). As the physiology of both the hybrid and pure *Mytilus edulis* is so similar there is likely to be very little impact on the tolerance of the bed to neither pressures nor a change in the associated fauna.

A review by Svåsand *et al*. (2007) concluded that there was a lack of evidence distinguishing between different populations to accurately assess the impacts of hybridisation and in particular how the gene flow may be affected by aquaculture. Therefore, it cannot be confirmed whether farming will have an impact on the genetics of this species beyond a potential for increased hybridisation.

The key characterizing species *Fucus serratus* is not currently cultivated or translocated. No information was found on current production of *Mastocarpus stellatus*, *Chondrus crispus* or other turf forming red seaweeds in the UK and it is understood that wild harvesting rather than cultivation is the method of production. No evidence was found for the effects of gene flow between cultivated species and wild populations. *Palmaria palmata* may be cultivated for use as biofilters and/or food. Experiments by van der Meer (1987) found that the hybrids of *Palmaria palmata* crosses from Canada and Ireland had vigorous growth and normal morphology, however the tetraspores had lower viability and those that germinated produced abnormal sporelings. It was concluded that populations from Ireland and Canada represent the same species that is in the process of splitting into sibling species. Populations around the UK express different haplotypes (Provan *et al*., 2005) indicating some genetic variation between populations. Although cultivation of different genotypes may lead to gene flow between wild and cultivated populations the limited dispersal may reduce exposure. Some negative effects may arise from hybridisation between very geographically separated populations but there is no evidence to suggest that gene flow between different UK haplotypes would lead to negative effects. Although cultivation of different genotypes may lead to gene flow between wild and cultivated populations the limited dispersal may reduce exposure.

Sensitivity assessment. No direct evidence was found regarding the potential for negative impacts of translocated mussel seed on adjacent natural beds. While it is possible that translocation of mussel seed could lead to genetic flow between cultivated beds and local wild populations, there is currently no evidence to assess the impact (Svåsand *et al*., 2007). Hybrid beds perform the same ecological functions as *Mytilus edulis* so that any impact relates to genetic integrity of a bed alone. This impact is considered to apply to all mussel bed biotopes equally, as the main habitat forming species *Mytilus edulis* is translocated. Also, given the uncertainty in identification of the species, habitats or biotopes described as dominated by *Mytilus edulis* may well be dominated by *Mytilus galloprovincialis*, their hybrids or a mosaic of the three. Presently, there is no evidence of impact due to genetic modification and translocation; therefore 'No evidence' is reported. The range of *Mytilus galloprovincialis* is thought to be extending northwards (Beaumont *et al*., 2007) and this assessment may require updating in the future. The pressure is considered to be 'Not relevant' to other species within the biotope.

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

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

Algal species which may have overlapping habitat requirements include the green seaweed *Codium fragile* subsp *tormentosoides* (now renamed as *Codium fragile fragile*) and the red seaweed *Heterosiphonia japonica*, neither of these have so far been recorded in nuisance densities (Sweet, 2011j). Wireweed, *Sargassum muticum*, grows best on sheltered shores and in rockpools (Sewell, 2011c), rather than the exposed shores characteristic of this biotope.The red seaweeds *Heterosiphonia japonica* and *Neosiphonia harveyi* may also occur in this biotope but again no impacts have been reported. The tunicates *Didemnum vexillum* and *Asterocarpa humilis*, the hydroid *Schizoporella japonica* and the bryozoan *Watersipora subatra* (Bishop, 2012c, Bishop, 2015a and b; Wood, 2015) are currently only recorded from artificial hard substratum in the UK and it is not clear what their established range and impacts in the UK would be.

A significant potential INIS is the Pacific oyster *Magallana gigas*, as its distribution and environmental tolerances are considered to overlap with this biotope and this reef forming species can alter habitat structure. This species may also affect the grazers present in the biotope. In the Wadden Sea and North Sea, *Magallana gigas* overgrows mussel beds in the intertidal zone (Diederich, 2005, 2006; Kochmann *et al*., 2008), although larvae did show preference for settling on conspecifics before the mussels and struggled to settle on mussels with a fucoid covering. It has been observed that mussel beds in the Wadden Sea that are adjacent to oyster farms were quickly converted to oyster beds (Kochmann *et al*., 2008). Padilla (2010) predicted that *Magallana gigas* could either displace or overgrown mussels on rocky and sedimentary habitats of low or high energy

Magallana gigas is the most widely grown bivalve in aquaculture around the world at present and an important nuisance species in marine waters (Padilla, 2010). Adults are also long-lived so that populations can survive with infrequent recruitment. It has a high fecundity, a long-lived pelagic larval phase and hence high dispersal potential (>1000km). *Magallana gigas* does not spawn at water temperatures below ca 20°C but adults grow in colder waters, so that it was thought that this species could not escape from cultivation in cold water areas. However, it has been suggested that climate change and warmer waters have allowed *Magallana gigas* to expand into and reproduce in previously unsuitable areas. Established feral populations have been reported to spread via larvae (Padilla, 2010). It is found form the mid-littoral to the upper subtidal, and grows on hard substrata but also on other bivalves (e.g. blue mussels) and polychaete reefs (Padilla, 2010).

Diederich (2005, 2006) examined settlement, recruitment and growth of *Magallana gigas* (as *Crassostrea gigas*) and *Mytilus edulis* in the northern Wadden Sea. *Magallana gigas* recruitment success was dependant on temperature, and in the northern Wadden Sea, only occurred in six of the 18 years since *Magallana gigas* was first introduced. Survival of juveniles is higher in mild than cold winters. Also survival of both juveniles and adults on mussel beds is higher than that of the mussels themselves. However, recruitment of *Magallana gigas* was significantly higher in the intertidal than the shallow subtidal, although the survival of adult oysters or mussels in the subtidal is limited by predation. Deiderich (2005) concluded that hot summers could favour *Magallana gigas* reproduction while cold winters could lead to high mussel recruitment the following summer. Diederich (2005, 2006) noted that the high survival rate of *Magallana gigas* adults and juveniles in the intertidal was likely to compensate for years of poor recruitment. *Magallana gigas* also prefer to settle on conspecifics, so that it can build massive oyster reefs, which themselves are more resistant of storms or ice scour than the mussel beds they replace; as oysters are cemented together, rather than dependent on byssus threads. *Magallana gigas* also grows faster than *Mytilus edulis* in the intertidal and reaches by ca 2-3 times the length of mussels within one year. In addition, growth rates in *Magallana gigas* were independent of tidal level (emergence regime, substratum, *Fucus* cover and barnacle epifauna (growing on both mussels and oysters), while growth rate of *Mytilus edulis* was decreased by these factors. The faster growth rate could make *Magallana gigas* more competitive than *Mytilus edulis* where space or food is limiting. Diederich (2006) concluded that the massive increase in *Magallana gigas* in the northern Wadden Sea was caused by high recruitment success, itself due to anomalously warm summer temperatures, the preference for settlement on conspecifics (and hence reef formation), and high survival rates of juveniles.   As oyster reefs form on former mussel beds, the available habitat for *Mytilus edulis* could be restricted (Diederich, 2006).

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

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

The non-native crab *Hemigrapsus sanguineus* has recently been recorded in the UK (Sweet & Sewell, 2014) and has the potential to be a significant predator of intertidal invertebrates. Significant reductions in common shore crab abundance and mussel density have been reported where the crab has achieved high densities in mainland Europe (Sweet & Sewell, 2014). Mortalities of 25% of juvenile *Mytilus edulis* were attributed to predation by *Hemigrapsus sanguineus* in an intertidal habitat of western Long Island Sound along the Connecticut coastline (Brousseau *et al*., 2014). In Rye, New York, declines of approximately 80% of *Littorina littorea* in the intertidal were reported to coincide with an expansion of the *Hemigrapsus sanguineus* population (Kraemer *et al*., 2007). This crab occurs on exposed shores and may therefore occur in this biotope when established. If predation of littorinids was significantly increased this could impact the algal composition and abundance of this biotope by altering the level of grazing pressure.

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

Introduction of microbial pathogens	Medium Q: High A: Medium C: Medium	Medium Q: High A: Low C: Medium	Medium Q: High A: Low C: Medium

Evidence for the impacts of microbial pathogens on *Mytilus edulis* was reviewed by Mainwaring *et al*. (2014) with specific reference to the shellfish pathogens *Marteilosis* and *Bonamia*. Natural *Mytilus edulis* beds are host to a diverse array of disease organisms, parasites and commensals from many animal and plant groups including bacteria, blue green algae, green algae, protozoa, boring sponges, boring polychaetes, boring lichen, the intermediary life stages of several trematodes, copepods and decapods (Bower, 1992; Gray *et al*., 1999; Bower, 2010). Whilst *Bonamia*, has been shown not to infect *Mytilus edulis* (Culloty *et al*., 1999), *Marteilia refringens* can infect and have significant impacts on the health of *Mytilus edulis*. Its distribution, impacts on the host, diagnostic techniques and control measures are reviewed by Bower (2011). There is some debate as to whether there are two species of *Marteilia*, one which infects oysters (*Marteilia refringens*) and another that infects blue mussels (*Marteilia maurini*) (Le Roux *et al*., 2001) or whether they are just two strains of the same species (Lopez-Flores *et al*., 2004; Balseiro *et al*., 2007). Both species are present in southern parts of the United Kingdom. The infection of *Marteilia* results in Marteiliosis which disrupts the digestive glands of *Mytilus edulis* especially at times of spore release. Heavy infection can result in a reduced uptake of food, reduced absorption efficiency, lower carbohydrate levels in the haemolymph and inhibited gonad development particularly after the spring spawning resulting in an overall reduced condition of the individual (Robledo *et al*., 1995).

Recent evidence suggests that *Marteilia* is transferred to and from *Mytilus edulis* via the copepod *Paracartia grani*. This copepod is not currently prevalent in the UK waters, with only a few records in the English Channel and along the South coast. However, it is thought to be transferred by ballast water and so localised introductions of this vector may be possible in areas of mussel seed transfer e.g. the Menai Strait. The mussel populations here are considered to be naive (i.e. not previously exposed) and therefore could be heavily affected, although the likelihood is slim due to the dependence on the introduction of a vector that is carrying *Marteilia* and then it being transferred to the mussels. Berthe *et al*. (2004) concluded that *Mytilus edulis* is rarely significantly affected by *Marteilia* sp. However, occasions have been recorded of nearly 100 % mortality when British spat have been transferred from a 'disease free area' to areas in France were *Marteilia sp*. are present. This suggests that there is a severe potential risk if naive spat are moved around the UK from northern waters into southern waters where the disease is resident (enzootic) or if increased temperatures allow the spread of *Marteilia sp.* northwards towards the naive northern populations. In addition, rising temperatures could allow increased densities of the *Marteilia sp.* resulting in heavier infections which can lead to mortality.

Very little is known about infections in *Fucus* (Wahl *et al.*, 2011). Coles (1958) identified parasitic nematodes that caused galls on *Fucus serratus* in the Southwest of Britain. But to date no mortalities have been associated to the introduction of microbial pathogens. Torchin *et al.*, (2002) suggests that there is potential for increased biotic interactions with parasites or pathogens in many marine systems. More recently, Zuccaro *et al.* (2008) detected a number of fungal species associated with *Fucus serratus*. So far no mortalities have been associated to the introduction of microbial pathogens. However, the potential for increased biotic interactions involving parasites or pathogens is on the rise in many marine systems (Torchin *et al.*, 2002). No evidence was found for pathogens of red algae which may be present in this biotope. The fungal pathogen, *Petersenia palmaria* n. sp (Oomycetes) which infects *Palmaria mollis*, does not affect *Palmaria palmata* (Meer & Pueschel, 1985).

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

Sensitivity assessment. No evidence was found that outbreaks of microbial pathogens significantly impact populations of the key characterizing *Fucus vesiculosus*and other associated algal species. Limpets, barnacles and littorinids may be subject to persistent low levels of infestation by pathogens but these are not recorded to lead to high-levels of mortality. Bower (2010) noted that although *Marteilia* was a potentially lethal pathogen of mussels, most populations were not adversely affected by marteilioisis but that in some areas mortality can be significant in mariculture (Berthe *et al.*, 2004). The resultant population would be more sensitive to other pressures, even where the disease only resulted in reduced condition. Therefore, a precautionary resistance of 'Medium' is suggested (<25 % mortality), with a resilience of 'Medium' (2-10 years) resulting in a sensitivity of 'Medium'.

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

Direct, physical impacts from harvesting are assessed through the abrasion and penetration of the seabed pressures. The sensitivity assessment for this pressure considers any biological/ecological effects resulting from the removal of target species on this biotope. Many of the species characterizing or associated with this biotope may be targeted by either recreational or commercial harvesters.

Mytilus edulis is a commercially targeted species worldwide and has been fished for hundreds of years and managed in England and Wales for the last hundred years (Holt *et al*., 1998). Mussels are also regularly hand collected by fisherman for bait and food from intertidal beds which can also result in significant damage to the bed (Holt *et al*., 1998; Smith & Murray, 2005). Smith & Murray (2005) examined the effects of low level disturbance and removal on an extensive bed of *Mytilus californianus* (composed of a single layer of mussels) in southern California. They observed a significant decrease in mussel mass (g/m^2), density (no/m^2), percentage cover and mean shell length due to low-intensity simulated bait-removal treatments (2 mussels / month) for 12 months (Smith & Murray 2005). They also stated that the initial effects of removal were ‘overshadowed’ by loss of additional mussels during time periods between treatments, probably due to the indirect effect of weakening of byssal threads attachments between the mussel leaving them more susceptible to wave action (Smith & Murray, 2005). The low-intensity simulated bait-removal treatments had reduced percentage cover by 57.5% at the end of the 12 month experimental period. Smith & Murray (2005) suggested that the losses occurred from collection and trampling are far greater than those that occur by natural causes. This conclusion was reached due to significant results being displayed for human impact despite the experiment taking place during a time of high natural disturbance from El Niño–Southern Oscillation (ENSO). In addition, Holt *et al*., (1998) recorded an incident of the removal of an entire bed that is adjacent to a road in Anglesey due to fishermen bait collecting. Recreational fishermen will often collect moulting *Carcinus maenas* or whelks by hand from intertidal mussel beds for bait. The removal of predatory crabs could actively benefit *Mytilus edulis*.

Stagnol *et al*. (2013) investigated the effects of commercial harvesting of intertidal fucoids on ecosystem biodiversity and functioning. The study found that the removal of the macroalgae canopy affected the metabolic flux of the area. Flows from primary production and community respiration were lower on the impacted area as the removal of the canopy caused changes in temperature and humidity conditions. Other studies confirm that loss of canopy had both short and long-term consequences for benthic communities in terms of diversity resulting in shifts in community composition and a loss of ecosystem functioning such as primary productivity (Lilley & Schiel, 2006; Gollety *et al*., 2008). Red algae within the biotope may also be subject to hand gathering. *Mastocarpus stellatus* is harvested commercially in Scotland and Ireland to produce carageen, the stipe is removed but the base is left intact to allow the algae to re-grow. *Palmaria palmata* and *Osmundea pinnatifida* are also collected by hand commercially and recreationally for consumption.

Littorinids are one of the most commonly harvested species of the rocky shore. Large scale removal of *Littorina littorea* may allow a proliferation of opportunistic green algae, such as *Ulva*, on which it preferentially feeds. Experiments designed to test the effects of harvesting by removing individuals at Strangford Lough found that there was no effect of experimental treatments (either harvesting or simulated disturbance) on *Littorina littorea* abundance or body size over a 12 week period (Crossthwaite *et al*. 2012). This suggests that these animals are generally abundant and highly mobile; thus, animals that were removed were quickly replaced by dispersal from surrounding, un-harvested areas. However, long-term exploitation, as inferred by background levels of harvest intensity, did significantly influence population abundance and age structure

(Crossthwaite *et al.* 2012). A broadscale study of harvesting in Ireland using field studies and interviews with wholesalers and pickers did suggest that some areas were over harvested but the lack of background data and quantitative records make this assertion difficult to test (Cummins *et al*., 2002). Changes in grazer abundance can alter the character of the algal assemblage. Grazer removal (manual removal of all gastropods in pool and a 1m surrounding perimeter) caused strong and highly significant changes in assemblage structure in rockpools that contained red turf forming algae mainly due to an increase in the cover of green filamentous algae and a decrease in cover of live crustose coralline algae (25.40%) (Atalah & Crowe, 2010).

Sensitivity assessment *Mytilus edulis* beds have no avoidance mechanisms to escape targeted harvesting and as a result a significant proportion of the bed can be removed (Palmer *et al*., 2007; Narvarte *et al*., 2011). As the majority of the mussel beds that are harvested in the UK are regularly replenished with seed, the recovery rate for maintained beds should be rapid. In natural (wild) beds, the recovery could be significantly longer due to indirect effects from wave action and the sporadic nature of recruitment (Paine & Levin 1981; Seed & Suchanek 1992). Even hand-picking for bait can result in a significant decrease in cover, especially in beds composed of a single layer of mussels (Smith & Murray 2005). It should be noted that dense, multi-layered mussel beds may be more resistant to the gaps and bait collection, as damage to the upper layer may not effect deeper layers, so that attachment to the substratum and each other is maintained (Brosnan & Crumrine, 1994). Based on the available evidence *Mytilus edulis* are considered to have 'Low' resistance to this pressure and 'Medium' resilience so that sensitivity is assessed as 'Medium'. As the other species that are harvested in this biotope are also attached, sedentary or slow moving and relatively conspicuous a single event of targeted harvesting could efficiently remove individuals and resistance is assessed as 'Low'. Resilience of *Fucus serratus*, the turf forming red seaweeds and littorinids is assessed as 'High' (based on evidence for recovery from harvesting that did not damage the algal bases although see caveats in the resilience section) and biotope sensitivity is assessed as 'Low'. This assessment refers to a single collection event, long-term harvesting over wide spatial scales will lead to greater impacts, with lower resistance and longer recovery times. Loss of *Mytilus edulis* would lead to reclassification of the biotope and the greater sensitivity of this species is presented in the sensitivity assessment table.

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

Direct, physical impacts from harvesting are assessed through the abrasion and penetration of the seabed pressures, while this pressure considers the ecological or biological effects of by-catch. Removal of a large part of the *Mytilus edulis* bed and the brown and red macroalgae would substantially alter the character of the biotope, leading to reclassification. Loss of the characterizing and associated species would also alter ecosystem functions, such as rates of production and the provision of a structurally complex habitat.

Sensitivity assessment. Removal of individuals as by-catch would remove the biological assemblage that defines the biotope, hence the biotope is considered to have 'Low' resistance to this pressure and to have 'Medium' resilience (based on *Mytilus edulis*). Biotope sensitivity is therefore 'Medium'.

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