



# MarLIN

## Marine Information Network

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

# *Sabellaria spinulosa* with kelp and red seaweeds on sand-influenced infralittoral rock

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

Jacqueline Hill, Dr Heidi Tillin, Charlotte Marshall & Natalie Gibb

2020-02-21

A report from:

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

**Please note.** This MarESA report is a dated version of the online review. Please refer to the website for the most up-to-date version [<https://www.marlin.ac.uk/habitats/detail/144>]. All terms and the MarESA methodology are outlined on the website (<https://www.marlin.ac.uk>)

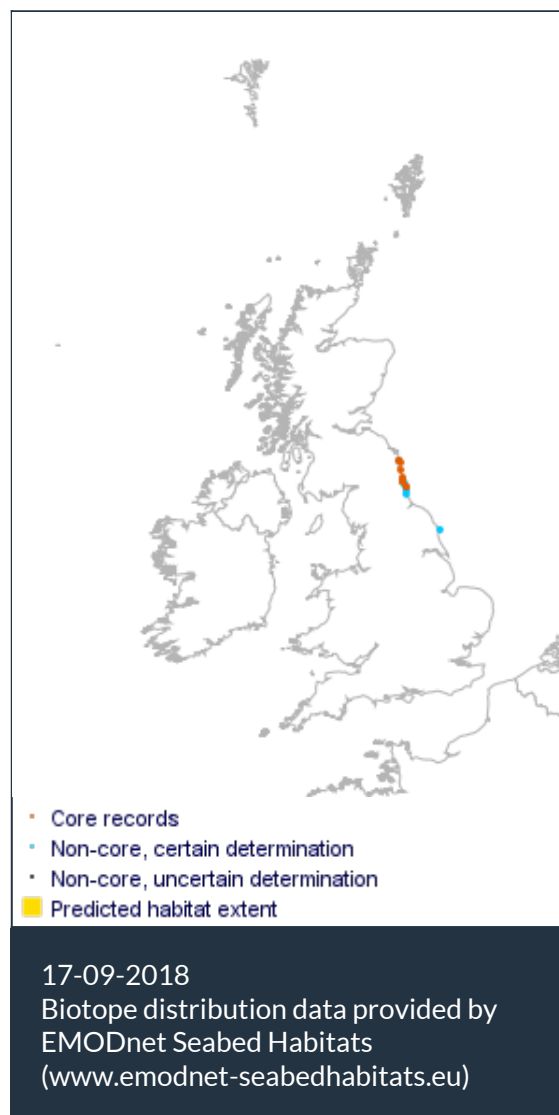
This review can be cited as:

Hill, J.M., Tillin, H.M., Marshall, C. & Gibb, N., 2020. [*Sabellaria spinulosa*] with kelp and red seaweeds on sand-influenced infralittoral rock. In Tyler-Walters H. and Hiscock K. (eds) *Marine Life Information Network: Biology and Sensitivity Key Information Reviews*, [on-line]. Plymouth: Marine Biological Association of the United Kingdom. DOI <https://dx.doi.org/10.17031/marlinhab.144.2>



The information (TEXT ONLY) provided by the Marine Life Information Network (MarLIN) is licensed under a Creative Commons Attribution-Non-Commercial-Share Alike 2.0 UK: England & Wales License. Note that images and other media featured on this page are each governed by their own terms and conditions and they may or may not be available for reuse. Permissions beyond the scope of this license are available [here](https://www.marlin.ac.uk). Based on a work at [www.marlin.ac.uk](https://www.marlin.ac.uk)

(page left blank)



**Researched by** Jacqueline Hill, Dr Heidi Tillin, Charlotte Marshall & Natalie Gibb

**Refereed by** This information is not refereed

## Summary

### ☰ UK and Ireland classification

<b>EUNIS 2008</b>	A3.2145	<i>Sabellaria spinulosa</i> with kelp and red seaweeds on sand-influenced infralittoral rock
<b>JNCC 2015</b>	IR.MIR.KR.Lhyp.Sab	<i>Sabellaria spinulosa</i> with kelp and red seaweeds on sand-influenced infralittoral rock
<b>JNCC 2004</b>	IR.MIR.KR.Lhyp.Sab	<i>Sabellaria spinulosa</i> with kelp and red seaweeds on sand-influenced infralittoral rock
<b>1997 Biotope</b>	IR.MIR.SedK.SabKR	<i>Sabellaria spinulosa</i> with kelp and red seaweeds on sand-influenced infralittoral rock

### 🔍 Description

*Laminaria hyperborea* kelp forest on shallow infralittoral bedrock and boulders characterized by

encrustations of *Sabellaria spinulosa* tubes that cover much of the rock, together with sand-tolerant red seaweeds such as *Phyllophora pseudoceranoides*, *Dilsea carnosa* and *Polysiphonia elongata* and *Polysiphonia fucooides*. Red seaweeds such as *Plocamium cartilagineum* and *Delesseria sanguinea* may also be found beneath the kelp canopy, although typically low in abundance. They can be colonized by the ascidian *Botryllus schlosseri*. The cowrie *Trivia arctica* can also be found here. Much of the available rock is covered with encrusting coralline algae together with patches of the encrusting sponge *Halichondria panicea* and the sea anemone *Urticina felina*. More mobile fauna include the echinoderms *Asterias rubens*, *Henricia sanguinolenta*, *Echinus esculentus* and *Ophiothrix fragilis*, the gastropod *Gibbula cineraria* and the hermit crab *Pagurus bernhardus*. The scouring effect of mobile sand adjacent to the rock maintains a reduced underflora and fauna compared to the association of species found in non-scoured kelp forests (Lhyp.Ft). Scour-resistant fauna such as the barnacle *Balanus crenatus* can be locally abundant on the rock, while the bivalve *Pododesmus patelliformis* can be found seeking shelter underneath the cobbles. Above the effect of scour, kelp stipes may be densely colonized by red seaweeds such as *Phycodrys rubens*, *Palmaria palmata* and *Membranoptera alata*, together with some sponges and ascidians. This biotope is found in the sand-laden waters of north-east England in conditions in which *Sabellaria spinulosa* is able to thrive. Nearby circalittoral rock is often also dominated by *Sabellaria spinulosa* (Sspi) but lacks the kelp and red seaweeds. This biotope is not commonly recorded in the UK so there is a lack of information relating to the surrounding biotopes. (Information from Connor *et al.*, 2004; JNCC, 2015).

### ↓ Depth range

0-5 m, 5-10 m

### Additional information

-

### ✓ Listed By

- none -

### Further information sources

Search on:



## Habitat review

### 🔄 Ecology

#### Ecological and functional relationships

- *Sabellaria spinulosa* colonize scoured rock rapidly and may be sufficiently dense to prevent the settlement or attachment of other species to the substratum, although the crust itself may act as a substratum for other fauna and flora.
- *Sabellaria spinulosa* requires suspended sand grains in order to form its tubes; reef communities therefore, only occur in turbid areas where sand is placed into suspension by water movement.
- Kelps are major primary producers, up to 90% of kelp production enters the detrital food web and kelp is probably a major contributor of organic carbon to surrounding communities (Birkett *et al.*, 1998b).
- Kelp fronds, stipes and holdfasts provide substrata for distinct communities of species, some of which are found only or especially on kelp plants. Kelp holdfasts provide both substrata and refugia for a huge diversity of macroinvertebrates. Kelp beds are diverse species rich habitats and over 1,800 species have been recorded in the UK kelp biotopes (Birkett *et al.*, 1998b).
- Epiphytes and understory algae are grazed by a variety of amphipods, isopods and gastropods, e.g. *Littorina* spp., *Acmaea* spp., *Haliotis tuberculata*, *Aplysia* and rissoid gastropods (Birkett *et al.*, 1998b).
- *Sabellaria spinulosa* and other associated organisms in the biotope, may be an important source of food for the pink shrimp *Pandalas montagui*. The biotope may also be an important feeding ground for fish.
- Suspension feeders, such as *Sabellaria spinulosa*, *Ophiothrix fragilis*, sponges, bryozoans and ascidians are the dominant fauna in the biotope. The top shell *Steromphala cineraria* is the only common grazer in the biotope although *Echinus esculentus* is also sometimes present. The anemone *Urticina felina* is a passive carnivore, waiting to trap animals that stumble into its tentacles.
- Although not present in large numbers in the biotope *Echinus esculentus* can have an influence in the biotope. The species graze the under-canopy and understory algae, including juvenile kelp sporophytes, together with epiphytes and epifauna on the lower reaches of the laminarian stipe. Wave action and abrasion between stipes probably knocks urchins off the upper stipe. It is likely that urchins will graze the *Sabellaria spinulosa*. Sea urchin grazing may maintain the patchy and species rich understory epiflora/fauna by preventing dominant species from becoming established.

#### Seasonal and longer term change

- *Sabellaria spinulosa* is a fast growing annual species and crusts up to 2-3cm thick can develop within one growing season. High recruitment of *Sabellaria spinulosa* may result in 'reinforcement' of the crust of tubes on the substratum. Reproductive seasonality of *Sabellaria spinulosa* is unclear, but spawning probably occurs largely over winter and settlement in early spring (Holt *et al.*, 1998).
- New blades of *Laminaria hyperborea* grow in winter between the meristem and the old blade, which is shed in early spring or summer together with associated species growing on its surface. Larger and older plants become liable to removal by wave action and storms due to their size and weakening by grazers such as *Patella pellucida*. Loss of older

plants results in more light reaching the understory, temporarily permitting growth of algae including *Laminaria hyperborea* sporelings.

- Many species of red algae are perennial exhibiting strong seasonal patterns of growth and reproduction. *Delesseria sanguinea*, for example, produces new blades in February and grows to full size by May - June becoming increasingly battered or torn and the lamina are reduced to midribs by December (Maggs & Hommersand 1993).
- Several other species, including hydroids, are annuals and abundance may show seasonal changes.

### Habitat structure and complexity

The crusts of *Sabellaria spinulosa* appear to have a considerable influence on community structure by providing a single species sheet that may be unstable for other species to attach to. The development of a diverse community may be dependent on space being made in the *Sabellaria* crust and other species settling on the rock. Diversity on crusts is not high. It might be that the richest communities occur where *Sabellaria* is not dominant. This is in contrast to *Sabellaria spinulosa* reefs on mobile substrata such as cobbles and pebbles which are stabilised by the crusts and often have a higher diversity and abundance of fauna than nearby areas (George & Warwick, 1985) with fauna such as sponges, ascidians, hydroids and bryozoans attached to the crust. The presence of kelp plants, and other algae, contribute to increases in structural complexity as the fronds, stipe and holdfast provide substratum and shelter for a great diversity and abundance of epiphytic algae and sessile fauna.

### Productivity

Productivity in the biotope is a mixture of primary and secondary productivity. Kelps are the major primary producers in UK marine coastal waters producing nearly 75 % of the net carbon fixed annually on the shoreline of the coastal euphotic zone (Birkett *et al.* 1998b). Kelp plants produce 2.7 times their standing biomass per year. Kelp detritus, as broken plant tissue, particles and dissolved organic material supports soft bottom communities outside the kelp bed itself. The kelps reduce ambient levels of nutrients, although this may not be significant in exposed sites, but increase levels of particulate and dissolved organic matter within the bed. However, kelp abundance, and hence productivity is not as high in the MIR.SabKR biotope as some other infralittoral biotopes (e.g. see EIR.LhypR). Many of the other species in the biotope, such as *Sabellaria spinulosa* and *Ophiothrix fragilis*, are suspension feeders feeding on detritus and phytoplankton.

### Recruitment processes

Most species present in the MIR.SabKR biotope possess a planktonic stage (gamete, spore or larvae) which float in the plankton before settling and metamorphosing into the adult form. This provides the potential for dispersal over considerable distances allowing many of the species in the biotope to rapidly colonize new areas that become available such as in the gaps often created by storms. The recruitment processes of key characteristic or dominant species are described here.

- Recruitment of *Sabellaria spinulosa* can be very variable. The larvae of *Sabellaria spinulosa* spend between six weeks and two months in the plankton (Wilson, 1970b) and so dispersal range is likely to be considerable. Larvae are strongly stimulated to settle by cement secretions of adult or newly settled individuals. In the absence of suitable stimulation metamorphosis and settlement occurs but always more slowly. High recruitment of *Sabellaria spinulosa* may result in 'reinforcement' of the crusts of tubes on

the substratum.

- *Laminaria hyperborea* produces vast numbers of spores, however they need to settle and form gametophytes within about 1 mm of each other to ensure fertilisation and therefore may suffer from dilution effects over distance. Gametophytes can survive darkness and develop in the low light levels under the canopy. However, young sporelings develop slowly in low light. Loss of older plants provides the opportunity to develop into adult plants. Most young sporophytes (germlings) appear in spring but can appear all year round depending on conditions (Birkett *et al.* 1998b).
- Rhodophyceae have non flagellate, and non-motile spores that stick on contact with the substratum. Norton (1992) noted that algal spore dispersal is probably determined by currents and turbulent deposition. However, red algae produce large numbers of spores that may settle close to the adult especially where currents are reduced by an algal turf or in kelp forests.
- Reproductive types of *Lithophyllum incrustans* occur from October to April but tail-off into summer. It has been calculated that 1 mm x 1mm of reproductive thallus produces 17.5 million bispores per year with average settlement of only 55 sporelings/year (Edyvean & Ford, 1984). However, spores will settle and new colonies will arise rapidly on bare substratum although growth rate is slow (2-7 mm per annum - see Irvine & Chamberlain, 1994).
- Some characterizing species may not recruit so readily, for instance the larvae of *Urticina felina* inhabits the water column, but is not considered to be truly pelagic and probably has limited dispersal abilities (Solé-Cava *et al.*, 1994).

### Time for community to reach maturity

*Sabellaria spinulosa* seems in many cases to act as a fast growing annual and early colonizer, but on more stable reefs the animals seem to be able to live for a few years. A typical lifespan for the littoral *Sabellaria alveolata* living in colonies forming reefs on bedrock in Duckpool was 4-5 years (Wilson, 1971). Areas where *Sabellaria spinulosa* had been lost due to winter storms appeared to recolonize quickly up to the maximum observed crust thickness (2.4 cm) during the following summer (R. Holt pers. comm. cited in Holt *et al.*, 1998). Linke (1951) reported that spawning of intertidal *Sabellaria spinulosa* reefs in the southern North Sea took place during the first and second years. Thus, in ideal conditions, sexual maturity of *Sabellaria spinulosa* is probably reached within the first year. The algae in the biotope are also likely to reach maturity fairly rapidly. Experimental clearance experiments in the Isle of Man (Kain 1975a; Kain, 1979) showed that *Laminaria hyperborea* returned to near control levels of biomass within 3 years at 0.8 m and the species reaches sexual maturity at between two and six years of age. Sivertsen (1991 cited in Birkett *et al.*, 1998b), showed that kelp populations stabilise about 4-5 years after harvesting. However, many of the other species, the anemone *Urticina felina* and coralline algae for example, within the reef matrix are slow growing and long-lived with a very low turnover rate. *Lithophyllum incrustans* in particular is very slow growing (2-7 mm per annum) and colonies may be up to 30 years old (Irvine & Chamberlain, 1994). Species diversity on the *Sabellaria* crust is likely to increase with age of the reef so although most components of the biotope can reach maturity within several years full community diversity and complexity is likely to take much longer.

### Additional information

-

## Preferences & Distribution

## Habitat preferences

<b>Depth Range</b>	0-5 m, 5-10 m
<b>Water clarity preferences</b>	No information
<b>Limiting Nutrients</b>	Nitrogen (nitrates), Phosphorus (phosphates)
<b>Salinity preferences</b>	Full (30-40 psu)
<b>Physiographic preferences</b>	Open coast
<b>Biological zone preferences</b>	Infralittoral
<b>Substratum/habitat preferences</b>	Bedrock, Large to very large boulders, Small boulders
<b>Tidal strength preferences</b>	Very Weak (negligible), Weak < 1 knot (<0.5 m/sec.)
<b>Wave exposure preferences</b>	Moderately exposed
<b>Other preferences</b>	Sand-scoured

## Additional Information

- No specific information is available regarding temperature preferences or tolerances for this biotope. The distribution of the key structural species *Sabellaria spinulosa* and *Laminaria hyperborea* extend to the north and south of the British Isles and so will be exposed to higher and lower temperatures than experienced locally.
- High levels of suspended sediment are likely to be required in order for *Sabellaria spinulosa* to construct its tubes.

## Species composition

### Species found especially in this biotope

### Rare or scarce species associated with this biotope

-

### Additional information



## Sensitivity review

### Sensitivity characteristics of the habitat and relevant characteristic species

This biotope IR.MIR.KR.Lhyp.Sab is characterized by *Laminaria hyperborea* kelp forest influenced by sand. The resultant scour reduces the abundance of red macroalgal and faunal turf understorey compared to typical kelp and red seaweed biotopes (KR) e.g. IR.MIR.KR.Lhyp.Ft. The suspended sediment also allows crusts of *Sabellaria spinulosa* to become abundant, further restricting substratum for foliose red seaweeds and faunal turf species, except for scour-resistant species such as *Balanus crenatus*, which may be locally abundant. However, the red algae, sponge and ascidian epiphytes typical of kelp stipes can occur above the level of surface scour. This biotope (KR.Lhyp.Sab) may represent a transition habitat between the infralittoral kelp biotopes and the circalittoral sand scour dominated *Sabellaria spinulosa* (CR.MCR.CSab.Sspi).

Loss of the kelp species would probably result in a faunal turf and *Sabellaria* crust dominated biotope, whereas loss of the scour and *Sabellaria* crusts would probably result in a typical *Laminaria hyperborea* dominated kelp forest biotope. Therefore, sensitivity assessment is based on the sensitivity of the two characteristic structural species, *Laminaria hyperborea* and *Sabellaria spinulosa*. The red algal flora and understorey would increase in abundance if the scour and *Sabellaria* crusts were lost, and the diversity of epiphytes is dependent on the presence of *Laminaria hyperborea* stipes. However, their sensitivity is mentioned where relevant. Other transient, opportunistic, or mobile species are probably ubiquitous and are not directly considered in the sensitivity assessment.

### Resilience and recovery rates of habitat

*Laminaria hyperborea* has a heteromorphic life strategy. A vast number of zoospores (mobile asexual spores) are released into the water column between October-April (Kain, 1964). Zoospores settle onto rock substrata and develop into dioecious gametophytes (Kain, 1979) which, following fertilization, develop into sporophytes and mature within 1-6 years (Kain, 1979; Fredriksen *et al.*, 1995; Christie *et al.*, 1998). *Laminaria hyperborea* zoospores have a recorded dispersal range of ~200 m (Fredriksen *et al.*, 1995). However zoospore dispersal is greatly influenced by water movements, and zoospore density and the rate of successful fertilization decreases exponentially with distance from the parental source (Fredriksen *et al.*, 1995). Hence, recruitment following disturbance can be influenced by the proximity of mature kelp beds producing viable zoospores to the disturbed area. (Kain, 1979, Fredriksen *et al.*, 1995).

Several review and experimental publications have assessed the recovery of *Laminaria hyperborea* kelp beds and the associated community. If environmental conditions are favourable *Laminaria hyperborea* can recover following disturbance events reaching comparable plant densities and size to pristine *Laminaria hyperborea* beds within 2-6 years (Kain, 1979; Birkett *et al.*, 1998b; Christie *et al.*, 1998). Holdfast communities may recover in six years (Birkett *et al.*, 1998b). Full epiphytic community and stipe habitat complexity regeneration require over six years (possibly 10 years). These recovery rates were based on discrete kelp harvesting events. Recurrent disturbance occurring frequently within 2-6 years of the initial disturbance is likely to lengthen recovery time (Birkett *et al.*, 1998b, Burrows *et al.*, 2014). Kain (1975a) cleared sublittoral blocks of *Laminaria hyperborea* at different times of the year for several years. The first colonizers and succession community differed between blocks and at the time of year the blocks were cleared, however within 2 years of clearance the blocks were dominated by *Laminaria hyperborea*.

In south Norway, *Laminaria hyperborea* forests are harvested, which results in large scale removal of the canopy-forming kelps. Christie *et al.* (1998) found that in south Norwegian *Laminaria hyperborea* beds a pool of small (<25 cm) understory *Laminaria hyperborea* plants persist beneath the kelp canopy for several years. The understory *Laminaria hyperborea* sporophytes had fully re-established the canopy at a height of one metre within 2-6 years after kelp harvesting. Within one year following harvesting, and each successive year thereafter, a pool of *Laminaria hyperborea* recruits had re-established within the understory beneath the kelp canopy. Christie *et al.* (1998) suggested that *Laminaria hyperborea* bed re-establishment from understory recruits (see above) inhibits the colonization of other kelps species and furthers the dominance of *Laminaria hyperborea* within suitable habitats, stating that *Laminaria hyperborea* habitats are relatively resilient to disturbance events.

*Laminaria hyperborea* biotopes are partially reliant on low (or no) populations of sea urchins, primarily the species *Echinus esculentus*, *Paracentrotus lividus* and *Strongylocentrotus droebachiensis*, which graze directly on macroalgae, epiphytes and the understory community. Multiple authors (Steneck *et al.*, 2002, 2004; Rinde & Sjøtun, 2005; Norderhaug & Christie, 2009; Smale *et al.*, 2013) have reported dense aggregations of sea urchins to be a principal threat to *Laminaria hyperborea* biotopes of the North Atlantic. Intense urchin grazing creates expansive areas known as 'urchin barrens', in which a shift can occur from *Laminaria hyperborea* dominated biotopes to those characterized by coralline encrusting algae, with a resultant reduction in biodiversity (Leinaas & Christie, 1996; Steneck *et al.*, 2002; Norderhaug & Christie, 2009). Continued intensive urchin grazing pressure on *Laminaria hyperborea* biotopes can inhibit the *Laminaria hyperborea* recruitment (Sjøtun *et al.*, 2006) and cause urchin barrens to persist for decades (Christie *et al.*, 1998; Steneck *et al.*, 2004; Rinde & Sjøtun, 2005). The mechanisms that control sea urchin aggregations are poorly understood but have been attributed to anthropogenic pressure on top-down urchin predators (e.g. cod or lobsters). Whereas these theories are largely unproven a number of studies have shown the removal of urchins from grazed areas coincide with kelp re-colonization (Leinaas & Christie, 1996; Norderhaug & Christie, 2009). Leinaas & Christie, (1996) removed *Strongylocentrotus droebachiensis* from "urchin barrens" and observed a succession effect, in which the substratum was initially colonized by filamentous macroalgae and *Saccharina latissima*. However, after 2-4 years *Laminaria hyperborea* dominated the community.

Reports of large scale urchin barrens within the North East Atlantic are generally limited to regions of the North Norwegian and Russian Coast (Rinde & Sjøtun, 2005, Norderhaug & Christie, 2009). Within the UK, urchin grazed biotopes (IR.MIR.KR.Lhyp.GzFt/Pk, IR.HIR.KFaR.LhypPar, IR.LIR.K.LhypSlat.Gz & IR.LIR.K.Slat.Gz) are generally localised to a few regions in North Scotland and Ireland (Smale *et al.*, 2013; Steneck *et al.*, 2002; Norderhaug & Christie 2009; Connor *et al.*, 2004). IR.MIR.KR.Lhyp.GzFt/Pk, IR.HIR.KFaR.LhypPar, IR.LIR.K.LhypSlat.Gz & IR.LIR.K.Slat.Gz are characterized by canopy-forming kelp. However, urchin grazing decreases the abundance and diversity of understory species. In the Isle of Man. Jones & Kain (1967) observed low *Echinus esculentus* grazing pressure can control the lower limit of *Laminaria hyperborea* and remove *Laminaria hyperborea* sporelings and juveniles. Urchin abundances in "urchin barrens" have been reported as high as 100 individuals/m<sup>2</sup> (Lang & Mann, 1978). Kain (1967) reported urchin abundances of 1-4 /m<sup>2</sup> within experimental plots of the Isle of Man. Therefore, whereas 'urchin barrens' are not presently an issue within the UK, relatively low urchin grazing has been found to control the depth distribution of *Laminaria hyperborea*, negatively impact on *Laminaria hyperborea* recruitment and reduce the understory community abundance and diversity.

Other factors that are likely to influence the recovery of *Laminaria hyperborea* biotopes is competitive interactions with Invasive Non-Indigenous Species (INIS), e.g. *Undaria pinnatifida*

(Smale *et al.*, 2013; Brodie *et al.*, 2014; Heiser *et al.*, 2014), and/or the Lusitanian kelp *Laminaria ochroleuca* (Brodie *et al.*, 2014; Smale *et al.*, 2015). A predicted sea temperature rise in the North and Celtic seas of between 1.5-5°C over the next century (Philippart *et al.*, 2011) is likely to create northward range shifts in many macroalgal species, including *Laminaria hyperborea*. *Laminaria hyperborea* is a northern (Boreal) kelp species, thus increases in seawater temperature is likely to affect the resilience and recoverability of *Laminaria hyperborea* biotopes with southerly distributions in the UK (Smale *et al.*, 2013; Stenneck *et al.*, 2002). Evidence suggests that the Lusitanian kelp *Laminaria ochroleuca* (Smale *et al.*, 2015), and the INIS *Undaria pinnatifida* (Heiser *et al.*, 2014) are competing with *Laminaria hyperborea* along the UK south coast and may displace *Laminaria hyperborea* from some subtidal rocky reef habitats. The wider ecological consequences of *Laminaria hyperborea*' competition with *Laminaria ochroleuca* and *Undaria pinnatifida* are however as of yet unknown.

Evidence to assess the likely recovery rate of *Sabellaria spinulosa* reefs from impacts is limited and significant information gaps regarding recovery rates, stability and persistence of *Sabellaria spinulosa* reefs exist (Gibb *et al.*, 2014). The use of evidence from different population densities e.g. between thin crusts and thick reefs and between *Sabellaria spinulosa* and the congener *Sabellaria alveolata* must, therefore, be treated cautiously as the evidence may not be applicable. 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 (Gibb *et al.*, 2014).

The reproductive phase appears to be relatively long and *Sabellaria spinulosa* larvae spend 6-8 weeks in the plankton (Wilson, 1970b). As a result, there is a good larval supply with high dispersal potential. Pearce *et al.* (2011a) found that separating the adult *Sabellaria spinulosa* from tubes in the laboratory, induced gamete release. Pearce *et al.* (2011a) suggested this was a 'significant evolutionary development whereby Sabellariid polychaetes spawn in response to disturbance as a means of potentially securing the future population'. Several studies have indicated that the major spawning event is in the spring. Plankton trawls revealed a high abundance of *Sabellaria spinulosa* larvae in February 2008 and smaller numbers in September and November 2009 (Pearce *et al.*, 2011a). Garwood (1982) found planktonic larvae on the north-east coast of England from August to November. These findings suggest that the main spawning event is at the beginning of the year but larvae are produced throughout the subsequent months. A February spawning event resulting in spring settlement is supported by the findings of George & Warwick (1985) and Wilson (1970a), who reported larval settlement in March in the Bristol Channel and Plymouth areas respectively. These findings suggest colonization of suitable habitats may be most likely in the spring but could occur over extended periods.

However, successful recruitment may be episodic. Wilson (1971) cites the work of Linke (1951) who recorded the appearance of *Sabellaria spinulosa* reefs on stone-work of intertidal protective groynes. In 1943 no colonies were present (time of year of this observation is unknown) but by September 1944 there were reefs 6-8 m wide and 40-60 cm high stretching for 60 m. Linke (1951) assumed that the settlement took place in 1944. In the summer of 1945 many colonies were dead and those remaining ceased growth in the autumn. Thick reefs may, therefore, develop rapidly and decline quickly. It should be noted, that these results should be interpreted cautiously, due to the possibility that the observed species may have been *Sabellaria alveolata* (Bryony Pearce, *pers comm.*).

The longevity of *Sabellaria spinulosa* reefs is not known and may vary between sites depending on

local habitat conditions. In naturally disturbed areas reefs may undergo annual cycles of erosion and recolonization (Holt *et al.*, 1998). Surveys on the North Yorkshire and Northumberland coasts found that areas where *Sabellaria spinulosa* had been lost due to winter storms, appeared to be recolonized up to the maximum observed 2.4 cm thickness during the following summer (R. Holt pers comm., cited from Holt *et al.*, 1998). Recovery of thin encrusting reefs may, therefore, be rapid.

Reefs may persist for long periods in some areas, although there is a significant lack of studies on the temporal stability of *Sabellaria spinulosa* reefs (Limpenny *et al.*, 2010). It has been suggested that the tubes of the worm can persist for some time in the marine environment. Therefore, the age of the colony may exceed the age of the oldest individuals present (Earll & Erwin, 1983). Laboratory experiments have suggested that larvae settle preferentially on old tubes (Wilson, 1970b). Therefore, providing environmental conditions are still favourable, the recovery of senescent or significantly degraded reefs through the larval settlement is stimulated by the presence of existing tubes (Earll & Erwin, 1983).

Studies of reefs of the congener *Sabellaria alveolata* within the low intertidal suggest that areas of small, surficial damage within reefs may be rapidly repaired by the tube building activities of adult worms. Vorberg (2000) found that trawl impressions made by a light trawl in *Sabellaria alveolata* reefs disappeared four to five days later due to the rapid rebuilding of tubes by the worms. Similarly, studies of intertidal reefs of *Sabellaria alveolata* by Cunningham *et al.* (1984) found that minor damage to the worm tubes as a result of trampling, (i.e. treading, walking or stamping on the reef structures) was repaired within 23 days. However, more severe, localised damage caused by kicking and jumping on the reef structure, resulted in large cracks between the tubes, and removal of sections (ca 15x15x10 cm) of the structure (Cunningham *et al.*, 1984). Subsequent wave action enlarged the holes or cracks. However, after 23 days, at one site, one side of the hole had begun to repair, and tubes had begun to extend into the eroded area. At another site, a smaller section (10x10x10 cm) was lost but, after 23 days, the space was already smaller due to rapid growth (Cunningham *et al.*, 1984). *Sabellaria spinulosa* reefs are more fragile than *Sabellaria alveolata* (Bryony Pearce, pers comm, 2014, cited in Gibb *et al.*, 2014) and recovery rates between reefs made by the two species may vary but this has not been established.

Where reefs are extensively damaged or removed, then recovery will rely on larval recolonization. *Sabellaria spinulosa* reproduction was studied by Wilson (1970a&b), Pearce *et al.* (2007) and Pearce *et al.* (2011b). Individuals may reach sexual maturity rapidly. Linke (1951) reported that *Sabellaria spinulosa* inhabiting the intertidal spawned at 1 or 2 years old and growth rate studies by Pearce *et al.* (2007) also suggested that sexual maturity in subtidal populations could be reached within the first year. Pearce *et al.* (2007) constructed size-frequency histograms based on wet weight of complete *Sabellaria spinulosa* collected from the Hastings Shingle Bank, which suggested that *Sabellaria spinulosa* was capable of rapid growth, approaching maximum adult biomass within months (Pearce *et al.*, 2007).

Studies within and adjacent to the Hastings Shingle Bank aggregate extraction area demonstrate a similarly rapid recolonization process (Cooper *et al.*, 2007; Pearce *et al.*, 2007). Recolonization within two previously dredged areas appeared to be rapid. Substantial numbers of *Sabellaria spinulosa* were recorded in one area in the summer after cessation of dredging activities and another area was recolonized within 16-18 months (Pearce *et al.*, 2007). Therefore, recruitment was annual rather than episodic in this area. Recovery to the high abundance and biomass of more mature reefs was considered to require 3-5 years if larval recruitment was successful every year (Pearce *et al.*, 2007).

However, in some cases reefs may not recover once removed. The Wadden Sea has experienced a widespread decline of *Sabellaria spinulosa* over recent decades with little sign of recovery. This is thought to be partly due to ecosystem changes (Reise *et al.*, 1989; Buhs & Reise, 1997) exacerbated by fishing pressures that continue (Riesen & Reise, 1982; Reise & Schubert, 1987). Likewise, no recovery of *Sabellaria spinulosa* has occurred in the approach channels to Morecambe Bay (Mistakidis 1956; cited from Holt *et al.*, 1998). This observation is believed this is due to a lack of larval supply or larval settlement, since larvae may preferentially settle on existing adult reefs (although directly settlement on sediments also occurs), or alterations in habitat (Holt *et al.*, 1998).

**Resilience assessment.** The evidence for recovery rates of *Sabellaria spinulosa* reefs from different levels of impact is very limited and the rates at which reefs recover from different levels of impact, and any similarity in rates or not between biotopes, have not been documented. Recovery rates are likely to be determined by a range of factors such as the degree of impact, the season of impact, larval supply and local environmental factors including hydrodynamics. The evidence from *Sabellaria alveolata* reefs (Vorberg, 2000; Cunningham *et al.*, 1984) suggests that areas of limited damage on a reef, e.g. where resistance is 'Medium', could be repaired rapidly (within weeks) through the tube-building activities of adults. It is not known if *Sabellaria spinulosa* exhibits the same response but resilience is assessed as 'High' in this instance. Predicting the rate of recovery following extensive removal is more problematic. Some thin crusts of *Sabellaria spinulosa* are relatively ephemeral and disappear following natural disturbance such as storms but recover the following year (Holt *et al.*, 1998), suggesting that recovery is 'High' (within 2 years), even where reefs are removed. In other instances, recolonization has been observed within 16-18 months. However, Pearce *et al.* (2007) suggested that full recovery to a state similar to the pre-impact condition of high adult density and adult biomass required three to five years where recruitment is annual. Therefore, recovery from significant impacts (where resistance is assessed as 'None' or 'Low') is probably 'Medium' (2-10 years).

The evidence suggests that beds of mature *Laminaria hyperborea* can regenerate from disturbance within 1-6 years, and the associated community within 7-10 years. However, other factors such as competitive interactions with *Laminaria ochroleuca* and *Undaria pinnatifida* may limit recovery of *Laminaria hyperborea* biotopes following disturbance. Also, urchin grazing pressure is shown to limit *Laminaria hyperborea* recruitment and reduce the diversity and abundance of the understory community and may limit habitat recovery following disturbance. The recovery of *Laminaria hyperborea* biotopes to disturbance from commercial harvesting in south Norway suggests that *Laminaria hyperborea* beds and the associated community could recover from a significant loss of canopy cover within 10 years. Therefore, resilience is probably 'Medium' (2-10 years).

**In summary**, the crusts of *Sabellaria spinulosa* that characterize this biotope may exhibit natural cycles of erosion and growth (e.g. after storms) and recover in the following year so that where the pressure under assessment results in 'some' damage to the crusts alone (a resistance of 'Medium') resilience is probably 'High'. However, where pressures result in 'some' damage to the kelp forest component of the biotope (a resistance of 'Medium') then resilience is probably 'Medium'. Nevertheless, where pressures result in 'significant' or 'severe' damage to the biotope (a resistance of 'Low' and 'None' respectively) then resilience is assessed as 'Medium' (2-10 years). The affected components of the biotope are highlighted where required. There is good evidence on the recovery of kelp dominated habitats but limited evidence on the recovery of *Sabellaria* crusts from impacts. Therefore, the confidence in the quality of the evidence varies between different components of the biotope. Therefore, confidence in the quality of the evidence is assessed as 'Medium'; the applicability of the evidence is also assessed as 'Medium' while the concordance is assessed as 'Medium' based on an agreement in direction but not magnitude, that

is, the rate of recovery between studies.

**Please note** as in Northern Norway urchin grazing pressure could extend the recovery of the *Laminaria hyperborea* biotopes >25 years. If intensive urchin grazing (as seen in Northern Norway) occurs in the UK resilience would be re-assessed as 'Very low'. However, because of the limited/localised incidence of urchin grazing within the UK, urchin grazing on large scales (as in Northern Norway) has not been included in this general resilience assessment. The introduction of Invasive Non-Indigenous Species (INIS) will also inhibit the recovery of *Laminaria hyperborea* biotopes for an indeterminate amount of time, in these cases, resilience would need to be re-assessed as 'Very low'. Another factor that is beyond the scope of this sensitivity assessment is the presence of multiple concurrent synergistic or cumulative effects, which Smale *et al.* (2013) suggest could be a more damaging than the individual pressures.

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

## Hydrological Pressures

	Resistance	Resilience	Sensitivity
Temperature increase (local)	Medium Q: High A: Medium C: Medium	Medium Q: Medium A: Medium C: Medium	Medium Q: Medium A: Medium C: Medium

Kain (1964) reported that *Laminaria hyperborea* sporophyte growth and reproduction could occur within a temperature range of 0°C and 20°C. Upper and lower lethal temperatures have been estimated at between 1-2°C above or below the extremes of this range (Birkett *et al.*, 1998b). Above 17°C gamete survival is reduced (Kain, 1964, 1971) and gametogenesis is inhibited at 21°C (Dieck, 1992). It is, therefore, likely that *Laminaria hyperborea* recruitment will be impaired at a sustained temperature increase of above 17°C. Sporophytes, however, can tolerate slightly higher temperatures of 20°C. Temperature tolerances for *Laminaria hyperborea* also vary seasonally and temperature changes are less tolerated in winter months than summer months (Birkett *et al.*, 1998b). However, it has a wide distribution in the North East Atlantic from Iceland south to the Bay of Biscay and Portugal.

Subtidal red algae are less tolerant of temperature extremes than intertidal red algae, surviving between -2°C and 18-23°C (Lüning 1990; Kain & Norton, 1990). A temperature increase may affect growth, recruitment or interfere with reproduction processes. For example, there is some evidence to suggest that blade growth in *Delesseria sanguinea* is delayed until ambient sea temperatures fall below 13°C. Blade growth is also likely to be intrinsically linked to gametangia development (Kain, 1987) and maintenance of sea temperatures above 13°C may affect recruitment success.

No empirical evidence was found for the temperature tolerance of *Sabellaria*

*spinulosa*. Nevertheless, its widespread distribution suggests that it is tolerant of temperature variation (Gibb *et al.*, 2014). *Sabellaria spinulosa* has the greatest geographical range of all the sabellariids, according to current records, encompassing Iceland, the Skagerrak and the Kattegat, the North Sea, the English Channel, the northeast Atlantic, the Mediterranean, the Wadden Sea and the Indian Ocean, (Achari, 1974; Riesen & Reise, 1982; Reise & Schubert, 1987; Hayward & Ryland, 1998; Foster-Smith, 2000; Collins, 2005). The associated epifauna are highly variable and reflect the assemblages found in adjacent biotopes.

**Sensitivity assessment.** In the UK, northern to southern Sea Surface Temperature (SST) ranges from 8-16°C in summer and 6-13°C in winter (Beszczynska-Möller & Dye, 2013). Overall, a chronic change (2°C for a year) outside the normal range for a year may reduce recruitment and growth, resulting in a minor loss in the population of kelp, especially in winter months or in southern examples of the biotope, although *Sabellaria* crusts are probably resistant. However, an acute change (5°C for a month; e.g. from thermal effluent) may result in loss of abundance of kelp or extent of the bed, especially in winter. Therefore, resistance to the pressure is assessed as 'Medium' and resilience as 'Medium' so that the sensitivity of this biotope to increases in temperature is assessed as 'Medium'.

#### Temperature decrease (local)

High

Q: High A: High C: Medium

High

Q: High A: High C: High

Not sensitive

Q: High A: High C: Medium

Kain (1964) reported that *Laminaria hyperborea* sporophyte growth and reproduction could occur within a temperature range of 0°C and 20°C. Upper and lower lethal temperatures have been estimated at between 1-2°C above or below the extremes of this range (Birkett *et al.*, 1998b). Subtidal red algae can survive at temperatures between -2°C and 18-23°C (Lüning, 1990; Kain & Norton, 1990). *Laminaria hyperborea* is a boreal northern species with a geographic range from mid-Portugal to Northern Norway (Birket *et al.*, 1998b), and a mid-range within southern Norway (60°-65° North)(Kain, 1971). The average seawater temperature for southern Norway in October is 12-13°C (Miller *et al.*, 2009). Also, the average annual sea temperature, from 1970 to 2014, was 8°C (Beszczynska-Möller & Dye, 2013). The available information suggests that *Laminaria hyperborea* forest and canopy would not be affected by a decrease in sea temperature at the benchmark level.

*Sabellaria spinulosa* occurs north to the Arctic and is, therefore, probably tolerant of a decrease in temperature at the pressure benchmark. This conclusion is supported by observations that *Sabellaria spinulosa* appeared unaffected by the cold, on oyster grounds in the River Crouch, throughout the severe winter of 1962–1963. The mean daily temperature was recorded at a depth of one fathom (1.8 m) below low water (equinoctial spring tide) and the lowest temperature recorded was -1.8°C (Crisp, 1964). At Penmon in Bangor, *Sabellaria spinulosa* also did not to suffer from the low temperatures and live individuals were found readily (Crisp, 1964).

**Sensitivity assessment.** Therefore, the evidence suggests that neither the *Laminaria hyperborea* canopy nor the *Sabellaria* crusts would be adversely affected by a decrease in temperature at the benchmark level. Therefore, resistance to the pressure is assessed as 'High' and resilience as 'High' so that the sensitivity of this biotope to a decrease in temperature is assessed as 'Not Sensitive' at the benchmark level.

#### Salinity increase (local)

Medium

Q: Low A: NR C: NR

Medium

Q: Medium A: Medium C: Medium

Medium

Q: Low A: Low C: Low



Lüning (1990) suggested that 'kelps' were stenohaline and that their general tolerance to salinity as a phenotypic group covered 16-50 psu over a 24 hr period. Optimal growth probably occurs between 30-35 psu ('Full' salinity) and growth rates are likely to be affected by periodic salinity stress. Birkett *et al.* (1998b) suggested that long-term increases in salinity may affect *Laminaria hyperborea* growth and may result in loss of affected kelp and, therefore, loss of the biotope. No evidence for the physiological tolerance of *Sabellaria spinulosa* to salinity change was found by Gibb *et al.* (2014). Roberts *et al.* (2010b) reported that hypersaline effluents from desalination plants disperse with tens of metres of the discharge point but reported widespread alteration in seagrass and soft sediment communities in poorly flushed environments. Hypersaline effluents are likely to sink to the seabed, and potentially penetrate the sediment. However, the water movement characteristic of this biotope is likely to disperse the effluent and limit the effect to the immediate vicinity of any discharge point.

**Sensitivity assessment.** The evidence suggests that the *Laminaria hyperborea* canopy might be affected by hypersaline conditions. Therefore, a precautionary resistance assessment of '**Medium**' is suggested but with Low confidence. Resilience is probably '**Medium**' so that sensitivity is assessed as '**Medium**'.

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

Lüning (1990) suggested that 'kelps' were stenohaline and that their general tolerance to salinity as a phenotypic group covered 16-50 psu over a 24 hr period. Optimal growth probably occurs between 30-35 psu ('Full' salinity) and growth rates are likely to be affected by periodic salinity stress. Birkett *et al.* (1998b) suggest that long-term changes in salinity may result in loss of affected kelp and, therefore, loss of this biotope.

Hopkin & Kain (1978) tested *Laminaria hyperborea* sporophyte growth at various low salinity treatments. The results showed that *Laminaria hyperborea* sporophytes could grow 'normally' at 19 psu, but that growth was reduced at 16 psu and sporophytes did not grow at 7 psu. A decrease in one MNCR salinity scale from 'Full' salinity (30-40 psu) to 'Reduced' salinity (18-30 psu) could result in a decrease of *Laminaria hyperborea* sporophyte growth. *Laminaria hyperborea* may also be out-competed by low salinity tolerant species e.g. *Saccharina latissima* (Karsten, 2007), or the Invasive Non-Indigenous Species *Undaria pinnatifida* (Burrows *et al.*, 2014). If salinity was returned to 'Full' salinity (30-40 psu) *Laminaria hyperborea* could out-compete *Saccharina latissima* and re-establish community dominance in 2-4 years (Kain, 1975; Leinaas & Christie, 1996), however full habitat structure may take over 10 years to recover (Birkett *et al.*, 1998b; Christie *et al.*, 1998). The ability of *Laminaria hyperborea* to out-compete *Undaria pinnatifida* within the UK is, however, unknown (Heiser *et al.*, 2014), and any interspecific interaction between *Laminaria hyperborea* and *Undaria pinnatifida* is not included within this sensitivity assessment.

No evidence for the physiological tolerances of *Sabellaria spinulosa* to decreases in salinity was found by Gibb *et al.* (2014). *Sabellaria spinulosa* does not seem to occur in very low salinity areas (Holt *et al.*, 1998) but has been recorded from estuaries including the Crouch, Mersey (Killeen & Light, 2000) and the Thames (Limpenny, 2010). Buhs & Reise (1997) surveyed 12 channel systems in the Wadden Sea and found that *Sabellaria spinulosa* reefs occurred in the northern tidal inlets that experienced salinity levels ranging from 28 to 30 psu. There is some speculation (Foster-Smith & Hendrick, 2003) that McIntosh (1922) misidentified samples of *Sabellaria spinulosa* as the congener *Sabellaria alveolata* from the Humber estuarine population (Holt *et al.*, 1998). These



records indicate that reduced and variable salinities can be tolerated to some extent but the paucity of records suggests that areas of reduced salinity do not provide optimal habitat.

**Sensitivity assessment.** Therefore a reduction in salinity from 'Full' (30-35) to 'Reduced' (18-30) for a year, may result in loss of a proportion of the kelp bed, as sporophyte growth is reduced and loss by wave action and herbivory increases, or via competition with opportunistic, low salinity tolerant, species such as *Saccharina latissima*. Similarly, *Sabellaria* may not be able to tolerate low salinities, although the evidence is unclear. Therefore, resistance to the pressure is assessed as 'Low', and resilience 'Medium' so that the sensitivity of this biotope to decreases in salinity is assessed as 'Medium'.

#### Water flow (tidal current) changes (local)

High

Q: High A: High C: Medium

High

Q: High A: High C: High

Not sensitive

Q: High A: High C: Medium

Kregting *et al.* (2013) measured *Laminaria hyperborea* blade growth and stipe elongation from an exposed and a sheltered site in Strangford Lough, Ireland, from March 2009 to April 2010. Maximal significant wave height (Hm0) was 3.67 & 2 m and maximal water velocity (Velrms) was 0.6 & 0.3 m/s at the exposed and sheltered sites respectively. Despite the differences in wave exposure and water velocity, there was no significant difference in *Laminaria hyperborea* growth between the exposed and sheltered sites. Therefore, water flow was found to have no significant effect on *Laminaria hyperborea* growth at the observed range of water velocities. Biotope structure is, however, different between wave exposed and sheltered sites. Pedersen *et al.* (2012) observed *Laminaria hyperborea* biomass, productivity and density increased with an increase in wave exposure. At low wave exposure, *Laminaria hyperborea* canopy-forming plants were smaller, had lower densities and had higher mortality rates than at exposed sites. At low wave exposure, Pedersen *et al.* (2012) suggested that high epiphytic loading on *Laminaria hyperborea* impaired light conditions, nutrient uptake, and increased the drag on the host *Laminaria hyperborea* during extreme storm events.

The morphology of the stipe and blade of kelps varies with water flow. In wave exposed areas, for example, *Laminaria hyperborea* develops a long and flexible stipe and this is probably a functional adaptation to strong water movement (Sjøtun, 1998). In addition, the lamina becomes narrower and thinner in strong currents (Sjøtun & Fredriksen, 1995). However, the stipe of *Laminaria hyperborea* is relatively stiff and can snap in strong currents. *Laminaria hyperborea* is usually absent from areas of high wave action or strong currents, although it is found in the Menai Strait, Wales, where tidal velocities can exceed 4 m/s (NBN, 2015) and in tidal rapids in Norway (J. Jones, pers. comm.). *Laminaria hyperborea* growth can persist in very strong tidal streams (>3 m/s). Increased water flow rate may also remove or inhibit grazers including *Patella pellucida* and *Echinus esculentus* and remove epiphytic algae growth (Pedersen *et al.*, 2012). The associated algal flora and suspension-feeding faunal populations change significantly with different water flow regimes. Increased water flow rates may reduce the understorey epiflora, to be replaced by an epifauna dominated community (e.g. sponges, anemones and polyclinid ascidians) as in the biotope IR.HIR.KFaR.LhypFa. The composition of the holdfast fauna may also change, e.g. energetic or sheltered water movements favour different species of amphipods (Moore, 1985).

*Sabellaria spinulosa* tends to occur in areas of high water movement where larvae, tube building materials and food particles are suspended and transported (Jones *et al.*, 2000). The relative importance of tidal versus wave induced movements to support reefs is, however, unclear (Holt *et al.*, 1998). There is limited in-situ data on the specific water flow tolerances of *Sabellaria spinulosa*,

although colonies have been found in areas with sedimentary bed forms that suggest current velocities in the range of 0.5 m/s to 1.0 m/s (Mistakidis, 1956; Jones *et al.*, 2000; Davies *et al.*, 2009). In the southern North Sea close to the coast of England, *Sabellaria spinulosa* reefs have been recorded in areas exposed to peak spring tidal flows of 1.0 m/s (Pearce *et al.*, 2014). Davies *et al.* (2009) also found, through laboratory experiments with *Sabellaria spinulosa* in tanks, that increasing the water flow to an average of 0.03 m/s is adequate to begin distribution of the sediment rain from the airlift throughout the tank and that doubling the water flow to almost 0.07 m/s further improved particle distribution throughout the tank. Therefore, it is likely that *Sabellaria spinulosa* requires habitats with a water flow above 0.07 m/s so that particles are suspended and distributed for the use of tube building and feeding.

Tillin (2010) used logistic regression to develop statistical models that indicate how the probability of occurrence of the congener *Sabellaria alveolata* changes over environmental gradients within the Severn Estuary. The model predicted response surfaces were derived for each biotope for each of the selected habitat variables, using logistic regression. From these response surfaces the optimum habitat range for each biotope could be defined based on the range of each environmental variable where the probability of occurrence, divided by the maximum probability of occurrence, is 0.75 or higher. These results identify the range for each significant variable where the habitat is most likely to occur. The modelled ranges should be interpreted with caution and apply to the Severn Estuary alone (which experiences large tidal ranges, high currents and extremely high suspended sediment loads and is therefore distinct from many other estuarine systems). However, these ranges do provide some useful information on environmental tolerances. The models indicate that for subtidal *Sabellaria alveolata* the maximum optimal current speed (the range in which it is most likely to occur) ranges from 1.26-2.46 m/s and the optimal mean current speed ranges from 0.5-1.22 m/s. Although not directly applicable to *Sabellaria spinulosa* this data suggests that tube-building Sabellariids can occur within a broad range of current speeds. In cases of reduced water flow, *Sabellaria spinulosa* is likely to suffer a reduction in the supply of suspended food and particles that are integral for growth and repair. A long-term decrease in water flow may reduce the viability of populations by limiting growth and tube building but no evidence was found for threshold levels relating to impact.

**Sensitivity assessment.** The IR.MIR.KR.Lhyp and CR.MCR.CSab biotopes and sub-biotopes are characteristic of moderate energy environments due to moderate or greater wave action or strong to weak tidal flow, depending on which source of water movement is dominant in any particular site. This biotope IR.MIR.KR.Lhyp.Sab biotope is found in weak tidal flow but moderate wave exposure, which keeps the sand in suspension. A significant decrease in water flow might reduce the suspension of sand to the detriment of the *Sabellaria* crusts but benefit red algae, whereas a significant increase in water flow may increase scour to the detriment of the red algal understorey and possibly kelp recruitment but benefit the *Sabellaria spinulosa*. Nevertheless, wave action is the dominant source of water movement in this biotope and a change in peak mean spring bed flow velocity of between 0.1m/s to 0.2m/s for more than 1 year is not likely to affect the structure of the biotope, especially as both kelp and red algal dominated biotopes and *Sabellaria* crusts occur in moderately strong (0.5-1.5 m/s) and strong water flows (1.5-3 m/s). Therefore, resistance to the pressure is assessed as 'High', and resilience 'High' so that the sensitivity of this biotope to changes in water flow is assessed as 'Not Sensitive' at the benchmark level.

#### Emergence regime changes

Low

Q: Low A: NR C: NR

Medium

Q: Medium A: Medium C: Medium

Medium

Q: Low A: Low C: Low

The upper limit of the *Laminaria hyperborea* bed is determined by wave action, water flow, desiccation, and competition from the more emergence resistant *Laminaria digitata*. *Laminaria hyperborea* exposed at extreme low water are very intolerant of desiccation, the most noticeable effect being bleaching of the frond and subsequent death of the meristem and loss of the plant. An increase in wave exposure (see below), as a result of increased emergence, has been found to exclude *Laminaria hyperborea* from shallow waters due to dislodgement of the sporophyte or snapping of the stipe (Birket *et al.*, 1998b). Hence, an increase in emergence could lead to mortality of exposed *Laminaria hyperborea* and the associated habitat. However, a decrease in emergence (at the benchmark level) may increase the upper depth restriction of *Laminaria hyperborea* forest biotope variants. However, limited light availability at depth will decrease the lower extent of *Laminaria hyperborea*, and may, therefore, result in a shift from forest to park biotope variants at depth. Further increases in depth will cause a community shift to that characterized by circalittoral faunal species, however, this is beyond the scope of the benchmark. *Sabellaria spinulosa* crusts are most abundant in the subtidal (e.g. CR.MCR.CSab) but crusts can occur in the lower intertidal.

**Sensitivity assessment.** Shallow examples of the biotope may lose a proportion of the *Laminaria* canopy due to an increase in emergence whereas an increase in emergence due to increase water depth might also limit the deeper extents of the *Laminaria* canopy. Therefore, resistance to the pressure is assessed as 'Low', and resilience 'Medium' so that the sensitivity of this biotope to changes in emergence is assessed as 'Medium'.

**Wave exposure changes (local)**

High

Q: Low A: NR C: NR

High

Q: High A: High C: High

Not sensitive

Q: Low A: Low C: Low

Kregting *et al.* (2013) measured *Laminaria hyperborea* blade growth and stipe elongation from an exposed and a sheltered site in Strangford Lough, Ireland from March 2009 to April 2010. Wave exposure was found to be between 1.1 to 1.6 times greater between the exposed and sheltered sites. Maximal significant wave height (Hm0) was 3.67 & 2 m and maximal water velocity (Velrms) was 0.6 & 0.3m/s at the exposed and sheltered sites respectively. Despite the differences in wave exposure and water velocity, there was no significant difference in *Laminaria hyperborea* growth between the exposed and sheltered site.

Biotope structure is, however, different between wave exposed and sheltered sites. Pedersen *et al.* (2012) observed *Laminaria hyperborea* biomass, productivity and density increased with an increase in wave exposure. At low wave exposure, *Laminaria hyperborea* canopy-forming plants were smaller, had lower densities and higher mortality rates than at exposed sites. At low wave exposure, high epiphytic loading on *Laminaria hyperborea* was theorised to impair light conditions, nutrient uptake, and increase the drag of the host *Laminaria hyperborea* during extreme storm events. The morphology of the stipe and blade of kelps vary with water flow. In wave exposed areas, for example, *Laminaria hyperborea* develops a long and flexible stipe and this is probably a functional adaptation to strong water movement (Sjøtun, 1998). In addition, the lamina becomes narrower and thinner in strong currents (Sjøtun & Fredriksen, 1995). However, the stipe of *Laminaria hyperborea* is relatively stiff and can snap in strong currents. *Laminaria hyperborea* is usually absent from areas of extreme wave action and can be replaced by *Alaria esculenta*. In extreme wave exposures, *Alaria esculenta* can dominate the shallow sublittoral to a depth of 15 m (Birket *et al.*, 1998b). Increase water flow rate may also remove or inhibit grazers including *Patella pellucida* and *Echinus esculentus* and remove epiphytic algae growth (Pedersen *et al.*, 2012). The associated algal flora and suspension-feeding faunal populations change significantly with different water flow regimes. Increased water flow rates may reduce the understory epiflora, to

be replaced by an epifauna dominated community (e.g. sponges, anemones and polyclinid ascidians) as in the biotope IR.HIR.KFaR.LhypFa. The composition of the holdfast fauna may also change, e.g. energetic or sheltered water movements favour different species of amphipods (Moore, 1985).

No direct evidence was found to assess this pressure on *Sabellaria spinulosa* crusts or reefs.

**Sensitivity assessment.** The IR.MIR.KR.Lhyp and CR.MCR.CSab biotopes and sub-biotopes are characteristic of moderate energy environments due to moderate or greater wave action or strong to weak tidal flow, depending on which source of water movement is dominant in any particular site. This biotope IR.MIR.KR.Lhyp.Sab biotope is found in weak tidal flow but moderate wave exposure, which keeps the sand in suspension. A significant decrease in wave action could reduce the suspension of sand to the detriment of the *Sabellaria* crusts but benefit red algae, whereas a significant increase in wave action could increase scour to the detriment of the red algal understorey and possibly kelp recruitment but benefit the *Sabellaria spinulosa*. Nevertheless, a change in nearshore significant wave height >3% but <5% is unlikely to have a significant effect in areas subject to moderate wave exposure. Therefore, resistance to the pressure is assessed as 'High', and resilience 'High' so that the sensitivity of this biotope to changes in water flow is assessed as 'Not Sensitive' at the benchmark level.

## ⚗ Chemical Pressures

	Resistance	Resilience	Sensitivity
<b>Transition elements &amp; organo-metal contamination</b>	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 <b>Not assessed</b> but evidence is presented where available.			
<b>Hydrocarbon &amp; PAH contamination</b>	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 <b>Not assessed</b> but evidence is presented where available.			
<b>Synthetic compound contamination</b>	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 <b>Not assessed</b> but evidence is presented where available.			
<b>Radionuclide contamination</b>	No evidence (NEv) Q: NR A: NR C:	Not relevant (NR) Q: NR A: NR C: NR	No evidence (NEv) Q: NR A: NR C: NR
No evidence			
<b>Introduction of other substances</b>	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 <b>Not assessed</b> but evidence is presented where available.			
<b>De-oxygenation</b>	High Q: Low A: NR C: NR	High Q: High A: High C: High	Not sensitive Q: Low A: Low C: Low

Reduced oxygen concentrations have been shown to inhibiting 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). Rapid recovery from a state of low oxygen is expected if the environmental conditions are transient. If levels do drop below 4 mg/l negative effects on these organisms can be expected with adverse effects occurring below 2 mg/l (Cole *et al.*, 1999). No information was found regarding *Sabellaria spinulosa* tolerance to changes in oxygenation.

**Sensitivity Assessment.** 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. Therefore, a resistance of 'High' is recorded. Resilience is assessed as 'High', and the biotope is assessed as 'Not sensitive' at the benchmark level.

<b>Nutrient enrichment</b>	<b>Not relevant (NR)</b>	<b>Not relevant (NR)</b>	<b>Not sensitive</b>
	Q: NR A: NR C: NR	Q: NR A: NR C: NR	Q: NR A: NR C: NR

Holt *et al.* (1995) suggest that *Laminaria hyperborea* may be tolerant of nutrient enrichment since healthy populations are found at ends of sublittoral untreated sewage outfalls in the Isle of Man. Increased nutrient levels e.g. from sewage outfalls have been associated with increases in abundance, primary biomass and *Laminaria hyperborea* stipe production but with concomitant decreases in species numbers and diversity (Fletcher, 1996). Increased nutrients may result in phytoplankton blooms that increase turbidity (see above). Increased nutrients may favour sea urchins, e.g. *Echinus esculentus*, due to their ability to absorb dissolved organics, and result in increased grazing pressure leading to loss of understorey epiflora/fauna, decreased kelp recruitment and possibly 'urchin barrens'. Therefore, although nutrients may not affect kelps directly, indirect effects such as turbidity, siltation and competition may significantly affect the structure of the biotope.

No direct evidence was found of the effects of nutrient enrichment on *Sabellaria spinulosa* crusts and reefs. However, this biotope is considered to be 'Not sensitive' at the pressure benchmark, that assumes compliance with good status as defined by the WFD.

<b>Organic enrichment</b>	<b>Medium</b>	<b>High</b>	<b>Low</b>
	Q: Medium A: Medium C: Medium	Q: Medium A: Medium C: Medium	Q: Medium A: Medium C: Medium

Holt *et al.* (1995) suggest that *Laminaria hyperborea* may be tolerant of organic enrichment since healthy populations are found at ends of sublittoral untreated sewage outfalls in the Isle of Man. Increased nutrient levels e.g. from sewage outfalls have been associated with increases in abundance, primary biomass and *Laminaria hyperborea* stipe production but with concomitant decreases in species numbers and diversity (Fletcher, 1996). Increase in ephemeral and opportunistic algae is associated with reduced numbers of perennial macrophytes (Fletcher, 1996). Increased nutrients may also result in phytoplankton blooms that increase turbidity. Therefore, although organic enrichment may not affect kelps directly, indirect effects such as turbidity may significantly affect the structure of *Laminaria hyperborea* biotopes.

*Sabellaria spinulosa* was reported to show enhanced growth adjacent to a sludge dumping area in Dublin Bay (Walker & Rees 1980). Hence, *Sabellaria spinulosa* reef biotopes are probably resistant to a high level of organic enrichment. Information on the levels of organic matter in Dublin Bay was not provided and so it is unclear how the levels experienced relate to the pressure benchmark. *Sabellaria spinulosa* reefs are found in areas of high water movement of up to 1 m/s



(Pearce *et al.*, 2014) that would naturally disperse some organic matter preventing accumulation and siltation. In larger, dense colonies of *Sabellaria spinulosa*, sand, detritus, and finer faecal materials collect in between worm tubes. These detritus layers do not interrupt the normal growth of the individuals or the colony as a whole (Schafer, 1972). Hence, it seems likely that *Sabellaria spinulosa* crusts and reefs are resistant to the deposition of a fine layer of organic materials.

Indirect effects arising from inputs of organic matter are possible where habitat quality and species interactions are altered. In the Wadden Sea, large subtidal areas of *Sabellaria spinulosa* reefs have been completely lost since the 1920s. This decline was partly attributed to an increase in coastal eutrophication that favoured blue mussel beds (Dörjes, 1992; Hayward & Ryland, 1998; Benson *et al.*, 2013). However, a direct causal link has not been established and it is possible that the decline of *Sabellaria spinulosa* reefs was due to physical damage from fishing activities rather than competitive interactions (Jones *et al.*, 2000).

**Sensitivity assessment.** Little evidence was found to support this sensitivity assessment. *Sabellaria spinulosa* and the associated species assemblage (which typically includes attached filter feeders from several phyla) is likely to be able to consume extra organic matter. This conclusion is supported by the enhanced growth rates recorded in the vicinity of sewage disposal areas (Walker & Rees, 1980). However, the *Laminaria* canopy and especially epiphytes and red algal epiflora may experience a reduction in abundance due to increased turbidity and competition with opportunistic green algae. Therefore, resistance is assessed as '**Medium**' as a precaution, albeit with 'Low' confidence. Hence, resilience is assessed as '**High**' and the sensitivity of this biotope as '**Low**'.

## A Physical Pressures

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

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

Physical change (to another seabed type)	<b>None</b> Q: High A: High C: High	<b>Very Low</b> Q: High A: High C: High	<b>High</b> Q: High A: High C: High
--	--	--	--

If rock substrata were replaced with sedimentary substrata this would represent a fundamental change in habitat type, which *Laminaria hyperborea* would not be able to tolerate (Birkett *et al.*, 1998b). *Sabellaria spinulosa* also develops crusts and reefs on coarse sediment and could probably recover. However, the biotope would no longer be a rock habitat, would lose its macroalgal component and be lost. Therefore, resistance to the pressure is assessed '**None**' and resilience '**Very Low**' so that sensitivity is 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 on hard rock habitats

## 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

## Not relevant on hard rock habitats

## Abrasion/disturbance of the surface of the substratum or seabed

Low

Q: High A: Medium C: Medium

Medium

Q: Medium A: Medium C: Medium

Medium

Q: Medium A: Medium C: Medium

Christie *et al.* (1998) observed *Laminaria hyperborea* habitat regeneration following commercial *Laminaria hyperborea* trawling in south Norway. Within the study area, trawling removed all large canopy-forming adult *Laminaria hyperborea*. However, sub-canopy recruits were largely unaffected. In 2-6 years of harvesting, a new canopy had formed one metre off the seabed. The associated holdfast communities recovered in six years, however, the epiphytic stipe community did not fully recover within the same period. Christie *et al.* (1998) suggested that kelp habitats were relatively resistant to direct disturbance/removal of *Laminaria hyperborea* canopy. Recurrent disturbance occurring at a smaller time scale than the recovery period of 2-6 years (stated above) could extend recovery time. Kain (1975) cleared sublittoral blocks of *Laminaria hyperborea* at different times of the year for several years. The first colonizers and succession community differed between blocks and the time of year the blocks were cleared. However, the blocks were dominated by *Laminaria hyperborea* within two years of clearance. Leinaas & Christie (1996) also observed *Laminaria hyperborea* re-colonization of 'urchin barrens', following the removal of urchins. The substratum was initially colonized by filamentous macroalgae and *Saccharina latissima* but *Laminaria hyperborea* dominated the community after 2-4 years.

*Sabellaria spinulosa* reef biotopes are directly exposed to physical damage that affects the surface. Gibb *et al.* (2014) found no direct evidence for impacts of the surface only for *Sabellaria spinulosa*. Studies of intertidal reefs of the congener *Sabellaria alveolata* (Cunningham *et al.*, 1984) found that the reef recovered within 23 days from the effects of trampling (i.e. treading, walking or stamping on the reef structures) by repairing minor damage to the worm tube porches. Severe damage caused by kicking and jumping on the reef structure, resulted in large cracks between the tubes, and removal of sections (ca 15 x15 x10 cm) of the structure (Cunningham *et al.*, 1984). Subsequent wave action enlarged the holes or cracks. However, after 23 days, at one site, one side of the hole had begun to repair, and tubes had begun to extend into the eroded area.

Vorberg (2000) used video cameras to study the effect of shrimp fisheries on *Sabellaria alveolata* reefs in the Wadden Sea. The imagery showed that a 3 m beam trawl easily ran over a reef that rose to 30 to 40 cm, although the beam was occasionally caught and misshaped on the higher sections of the reef. At low tide, there were no signs of the reef being destroyed and, although the trawl had left impressions, all traces had disappeared four to five days later due to the rapid rebuilding of tubes by the worms. The daily growth rate of the worms during the restoration phase was significantly higher than undisturbed growth, that is, the growth rate of undisturbed tubes was 0.7 mm/day but after removal of 2 cm of surface it was 4.4 mm/day, which indicated that as long as the reef is not completely destroyed recovery can occur rapidly.

*Sabellaria spinulosa* reefs are thought to be more fragile than *Sabellaria alveolata* (B. Pearce, *pers. comm.*, cited from Gibb *et al.*, 2014) and, therefore, surface abrasion may lead to greater damage and lower recovery rates than observed for *Sabellaria alveolata*. *Sabellaria spinulosa* reefs are often

only approximately 10 cm thick, surface abrasion can, therefore, severely damage and/or remove a reef. No direct observations of reef recovery, through repair, from abrasion were found for *Sabellaria spinulosa*.

**Sensitivity assessment.** Based on the evidence discussed above, abrasion at the surface of *Sabellaria spinulosa* crusts is likely to damage the tubes and result in sub-lethal and lethal damage to the worms. It is also likely to remove a proportion of the *Laminaria* canopy, attached epiphytes, *Laminaria* holdfasts and understorey macroalgae (where present). Therefore, resistance is assessed as 'Low' (loss of 25-75% of the extent/abundance of component species within the impact footprint). Hence, resilience is assessed as 'Medium' (within 2-10 years) and sensitivity as 'Medium'.

<b>Penetration or disturbance of the substratum subsurface</b>	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
<b>Changes in suspended solids (water clarity)</b>	Low Q: High A: High C: Medium	Medium Q: Medium A: Medium C: Medium	Medium Q: Medium A: Medium C: Medium

Not relevant on hard rock habitats, please refer to pressure 'Abrasion/disturbance of the substratum on the surface of the seabed' above.

Suspended Particle Matter (SPM) concentration has a linear relationship with subsurface light attenuation ( $K_d$ ) (Devlin *et al.*, 2008). An increase in SPM results in a decrease in sub-surface light attenuation. Light availability and water turbidity are principal factors that determine the depth range of *Laminaria hyperborea* (0-47 m Below Sea Level) (Birkett *et al.*, 1998b). Light penetration influences the maximum depth at which kelp species can grow and it has been reported that laminarians grow at depths at which the light levels are reduced to one per cent of incident light at the surface. Maximal depth distribution of laminarians, therefore, varies from 100 m in the Mediterranean to only 6-7 m in the silt-laden German Bight. In Atlantic European waters, the depth limit is typically 35 m. In very turbid waters, the depth at which *Laminaria hyperborea* is found may be reduced to 2.5 m (Birkett *et al.* 1998b), or in some cases excluded completely (e.g. Severn Estuary), because of the alteration in light attenuation by suspended sediment (Birkett *et al.* 1998b; Lüning, 1990).

*Laminaria* spp. show a decrease of 50% photosynthetic activity when turbidity increases by 0.1 /m (light attenuation coefficient =0.1-0.2 /m; Staehr & Wernberg, 2009). An increase in water turbidity will likely affect the photosynthetic ability of *Laminaria hyperborea* and decrease its abundance and density (see IR.HIR.KFaR.LhypR.Pk). Kain (1964) suggested that early *Laminaria hyperborea* gametophyte development could occur in the absence of light. Furthermore, observations from south Norway found that a pool of *Laminaria hyperborea* recruits could persist growing beneath *Laminaria hyperborea* canopies for several years, indicating that sporophytes growth can occur in light-limited environments (Christie *et al.*, 1998). However, in habitats exposed to high levels of suspended silts, *Laminaria hyperborea* is out-competed by *Saccharina latissima*, a silt tolerant species. Thus, a decrease in water clarity is likely to decrease the abundance of *Laminaria hyperborea* in the affected area (Norton, 1978). The biotope is expected to be excluded from silt rich environments.

*Sabellaria spinulosa* do not rely on light penetration for photosynthesis, it is also believed that visual perception is limited and that this species does not rely on sight to locate food or other resources. In a recent review of the sensitivity of *Sabellaria spinulosa* reefs to anthropogenic disturbance,



Fariñas-Franco *et al.* (2014) concluded that impacts on *Sabellaria spinulosa* due to a decrease in water clarity resulting from an increase in suspended solids (inorganic or organic) are unlikely, although no thresholds regarding tolerance or intolerance were found. Decreases in suspended particles that reduce the supply of food or tube-building materials may, however, negatively impact this species (Davies *et al.*, 2009; Last *et al.*, 2011).

*Sabellaria spinulosa* relies on a supply of suspended solids and organic matter to filter feed and build protective tubes and so they are often found in areas with high levels of turbidity. Davies *et al.* (2009) and Last *et al.* (2011) developed Vortex Resuspension Tanks (VoRT) to test the effects of a change in the composition of suspended sediment on benthic species. This laboratory experiment manipulated turbidity and current flow and demonstrated the susceptibility of *Sabellaria spinulosa* to a decrease in suspended particulate matter (SPM). A clear erosion of tubes was observed in the absence of SPM and subsequent starvation of tube building materials. At high and intermediate sediment regimes (high SPM ~71 mg/l) conditions were comparable to what might be expected within only a few hundred meters distance of a primary aggregate extraction site and *Sabellaria spinulosa* maintained a cumulative growth rate at these rates of SPM. This supports the view that the availability of suspended particles is necessary for *Sabellaria spinulosa* development and that tolerance of elevated levels is likely (Davies *et al.*, 2009). Indirect evidence for the tolerance of *Sabellaria spinulosa* for changes in turbidity is provided by the persistence of reefs on the outskirts of aggregate dredging areas (Pearce *et al.*, 2007, 2011a), which appear unaffected by dredging that is likely to have led to sediment plumes. Such plumes, however, are short-lived (Tillin *et al.*, 2011) and, therefore, the long-term effect depends on the duration of dredging activities.

Tillin (2010) used logistic regression to develop statistical models that indicate how the probability of occurrence of the congener *Sabellaria alveolata* changes over environmental gradients within the Severn Estuary. The modelled ranges should be interpreted with caution and apply to the Severn Estuary alone (which experiences large tidal ranges, high currents and extremely high suspended sediment loads and is therefore distinct from many other estuarine systems). The models indicate that the optimal mean neap sediment concentrations for subtidal *Sabellaria alveolata* range from 515.7 to 906 mg/l and optimal mean spring sediment concentrations range from 855.3 to 1631 mg/l. Although not directly applicable to *Sabellaria spinulosa* this data suggests that tube-building sabellariids are tolerant to very high levels of suspended sediment. Fine sediments such as mud may clog the gills and feeding tentacles of some polychaetes and, therefore, the potential impact will be mediated by the character of the sediment in suspension.

**Sensitivity assessment.** The benchmark for this pressure refers to a change in turbidity of one rank, e.g. from clear (<10 mg/l) to intermediate (10-100 mg/l) or intermediate to medium (100-300 mg/l). *Sabellaria spinulosa* do not photosynthesise and do not rely on sight to locate resources and therefore no effects are predicted for reef biotopes from an increase or decrease in clarity resulting from a change in one rank on the water framework directive scale. Experiments (Davies *et al.*, 2009) and predictive modelling (Tillin, 2010) indicate that tube building sabellariids can tolerate a broad range of suspended solids so the Sabellaria crusts are unlikely to be affected by an increase in suspended sediment. However, if the supply of sand to the habitat was reduced, then the *Sabellaria* crusts would probably erode and be replaced with a diverse understorey of red algae, similar to IR.MIR.KR.Lhyp. Similarly, an increase in turbidity will reduce the abundance of the *Laminaria hyperborea* canopy, especially in deeper examples of the biotope, and the *Laminaria* may be replaced by *Saccharina latissima* or by an abundant *Sabellaria* crust similar to CR.MCR.CSab.Sspi. Overall, the biotope is likely to change in character, be reclassified, and, effectively, lost. Therefore, resistance is assessed as 'Low', resilience as 'Medium', and sensitivity is assessed as 'Medium'.

**Smothering and siltation rate changes (light)****High**

Q: High A: High C: Medium

**High**

Q: High A: High C: High

**Not sensitive**

Q: High A: High C: Medium

Smothering by sediment (e.g. 5 cm of fine material) in a discrete event is unlikely to damage *Laminaria hyperborea* sporophytes but is likely to affect gametophyte survival as well as holdfast fauna, and interfere with zoospore settlement. Given the microscopic size of the gametophyte, 5 cm of sediment could be expected to significantly inhibit growth. However, laboratory studies showed that gametophytes can survive in darkness for between 6 and 16 months at 8°C and would probably survive smothering by a discrete event. Once returned to normal conditions the gametophytes resumed growth or maturation within one month (Dieck, 1993). Intolerance to this factor is likely to be higher during the peak periods of sporulation and/or spore settlement. If inundation is long-lasting then the understorey epifauna/flora may be adversely affected, e.g. suspension or filter-feeding fauna and/or algal species. However, this biotope is found in sand-laden waters (JNCC, 2015) so that the understorey of red algae and epifauna is limited in abundance.

*Sabellaria spinulosa* is often found in areas of high water movement with some degree of sediment transport essential for tube-building and feeding. *Sabellaria spinulosa* reefs adjacent to aggregate dredging areas appear unimpacted by dredging operations (Pearce *et al.*, 2007; Pearce *et al.*, 2011a). Evidence suggests that given the dynamic sedimentary environments in which sabellariids live, their populations can certainly persevere in turbid conditions despite 'typical' natural levels of burial (Last *et al.*, 2011) and that recovery from burial events is high. Last *et al.* (2011) buried *Sabellaria spinulosa* worms (isolated into artificial tubes), under three different depths of sediment – shallow (2 cm), medium (5 cm) and deep (7 cm). The results indicated that *Sabellaria spinulosa* could survive short-term (32 days), periodic, burial by sand of up to 7 cm. Last *et al.* (2011) suggested that the formation of 'emergence tubes' (newly created tubes extending to the surface) under sediment burial allowed *Sabellaria spinulosa* to tolerate gradual burial and that perhaps this mechanism allows for continued adult dispersal. This mechanism occurred most rapidly throughout the 8-day burial at ~1 mm per day (Last *et al.*, 2011). But even though tube-growth still seems possible under burial, the dumping of fine and coarse material will probably block feeding apparatus and, therefore, worm development will be curtailed.

**Sensitivity assessment.** This biotope occurs in moderately wave exposed conditions so that deposited sediments are unlikely to remain for more than a few tidal cycles and the effects of a deposit of 5 cm of fine sediment in a discrete event are likely to be transient. In addition, both *Laminaria hyperborea* and *Sabellaria spinulosa* crusts are unlikely to be adversely affected by a light deposition of 5 cm of sediment. Therefore, resistance to the pressure is assessed as 'High', and resilience as 'High' so that sensitivity is assessed as 'Not Sensitive' at the benchmark level.

**Smothering and siltation rate changes (heavy)****Low**

Q: High A: High C: Medium

**Medium**

Q: Medium A: Medium C: Medium

**Medium**

Q: Medium A: Medium C: Medium

Smothering by sediment (e.g. 30 cm of fine material) in a discrete event is unlikely to damage *Laminaria hyperborea* plants but is likely to affect gametophyte survival, holdfast communities, the epiphytic community at the base of the stipe, and interfere with zoospore settlement. Given the microscopic size of the gametophyte, 30 cm of sediment could be expected to significantly inhibit growth. However, laboratory studies showed that gametophytes can survive in darkness for between 6 and 16 months at 8°C and would probably survive smothering within a discrete event. Once returned to normal conditions the gametophytes resumed growth or maturation within 1

month (Dieck, 1993). Intolerance to this factor is likely to be higher during the peak periods of sporulation and/or spore settlement. If clearance of deposited sediment occurs rapidly then understorey communities are expected to recover quickly. If inundation is long-lasting then the understorey epifauna/flora may be adversely affected, e.g. suspension or filter-feeding fauna and/or algal species. However, this biotope is found in sand-laden waters (JNCC, 2015) so that the understorey of red algae and epifauna is limited in abundance.

*Sabellaria spinulosa* is often found in areas of high water movement with some degree of sediment transport essential for tube-building and feeding. *Sabellaria spinulosa* reefs adjacent to aggregate dredging areas appear unimpacted by dredging operations (Pearce *et al.*, 2007; Pearce *et al.*, 2011a). Evidence suggests that the dynamic sedimentary environments in which sabellariids live, their populations can persevere in turbid conditions in spite of 'typical' natural levels of burial (Last *et al.*, 2011) and that recovery from burial events is high. The congener *Sabellaria alveolata* was reported to survive short-term burial for days and even weeks in the south-west as a result of storms that altered sand levels up to two meters, although they were, killed by longer-term burial (Earll & Erwin 1983).

Last *et al.* (2011) buried *Sabellaria spinulosa* worms (isolated into artificial tubes), under three different depths of sediment – shallow (2 cm), medium (5 cm) and deep (7 cm). The results indicate that *Sabellaria spinulosa* can survive short-term (32 days), periodic burial by sand of up to 7 cm. Last *et al.* (2011) suggested that the formation of 'emergence tubes' (newly created tubes extending to the surface) under sediment burial allowed *Sabellaria spinulosa* to tolerate gradual burial and that perhaps this mechanism allows for continued adult dispersal. This mechanism occurred most rapidly throughout the 8-day burial at ~1 mm per day (Last *et al.*, 2011) but even though tube-growth still seems possible under burial, it is likely that the dumping of fine and coarse material will block feeding apparatus and, therefore, worm development will be curtailed.

A *Sabellaria spinulosa* reef off the coast of Dorset showed periodic burial from large sand waves (Collins, 2003). The displacement of some colonies that had established themselves on a gas pipeline 1 km off the coast of Aberdeen was also associated with burial (Mistakidis, 1956; cited by Holt *et al.*, 1998). Furthermore the loss of a 2 km<sup>2</sup> area of Ross worm reef in Jade Bay, North Sea was attributed to burial as a consequence of mud deposition, although fishing activity may have contributed to the decline (Dörjes, 1992, cited from Hendrick *et al.*, 2011). The evidence above suggests that *Sabellaria spinulosa* reefs are sensitive to damage from siltation events (Hendrick *et al.*, 2011). However, recovery is likely to be rapid given that larval dispersal is not interrupted and new reefs are likely to be able to establish themselves over old buried ones as postulated by (Fariñas-Franco *et al.*, 2014).

**Sensitivity assessment.** No direct evidence was found for the length of time that *Sabellaria spinulosa* can survive beneath 30 cm of sediment. Although this biotope occurs in moderate energy habitats (due to wave action), deposition of 30 cm of sediment represents a large volume of material that would probably remain for a number of tidal cycles and is expected to damage understorey flora/fauna as well as juvenile *Laminaria hyperborea*. Similarly, burial to *Sabellaria* crusts to a depth 30 cm of fine sediment may also result in loss the abundance or extent of the crusts, based on the observations in Hendrick *et al.* (2011). Therefore, as a precautionary assessment, resistance is assessed as 'Low' due to the depth of overburden. Hence, resilience is assessed as 'Medium' (2-10 years) and sensitivity as 'Medium'.

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.

<b>Electromagnetic changes</b>	No evidence (NEv) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	No evidence (NEv) Q: NR A: NR C: NR
No evidence			
<b>Underwater noise changes</b>	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			
<b>Introduction of light or shading</b>	<b>Low</b> Q: Low A: NR C: NR	<b>Medium</b> Q: Medium A: Medium C: Medium	<b>Medium</b> Q: Low A: Low C: Low

Shading of the biotope (e.g. by construction of a pontoon, pier etc) could adversely affect the biotope in areas where the water clarity is also low, and tip the balance to shade tolerant species, resulting in the loss of the biotope directly within the shaded area, or a reduction in laminarian abundance from forest to park type biotopes.

**Sensitivity assessment.** Resistance is probably 'Low', with a 'Medium' resilience and a sensitivity of 'Medium', albeit with 'low' confidence due to the lack of direct evidence.

<b>Barrier to species movement</b>	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR
------------------------------------	--	--	--

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

<b>Death or injury by collision</b>	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.** Collision from grounding vessels is addressed under abrasion above.

<b>Visual disturbance</b>	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
<b>Genetic modification &amp; translocation of indigenous species</b>	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

None of the important structuring characteristic species (e.g. *Laminaria hyperborea* or *Sabellaria spinulosa*) are subject to genetic modification or translocation, at present. Therefore, this pressure is considered 'Not relevant'.

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

The oyster drill, *Urosalpinx cinerea*, is not known to predate on polychaetes (Brown & Richardson,

1988), therefore, their introduction is not considered a threat to *Sabellaria spinulosa*. There is, however, some overlap between the environmental niche of *Sabellaria spinulosa* and the oysters that *Urosalpinx cinerea* selectively feed on (Brown & Richardson, 1988). Japanese wireweed *Sargassum muticum* and green sea fingers *Codium fragile* have the potential to compete for space where *Sabellaria spinulosa* reefs occur intertidally, however, intertidal biotopes are not included in this assessment and these species are unlikely to impact deeper subtidal reefs. The assessed biotope occurs in areas of high turbidity and the associated fauna tend to be scour tolerant so that wireweed and green sea fingers would probably be excluded.

No records of the carpet sea squirt *Didemnum vexillum*, the reef building serpulid *Ficopomatus enigmaticus*, the colonial sea squirt *Perophora japonica*, or Japanese kelp *Undaria pinnatifida*, suggest these species occur on or near *Sabellaria spinulosa* reefs. However, the further spread may impact subtidal *Sabellaria spinulosa* reefs through smothering or competition, although this is entirely speculative. The assessed biotope supports a variety of attached epifauna including species of bryozoans, hydroids and sponges so the stalked sea squirt *Styela clava* may be able to colonize this biotope but is unlikely to have greater negative impacts than native species.

Two species that potentially pose a threat to *Sabellaria spinulosa* reefs or crusts are the Pacific oyster *Magallana gigas* and the slipper limpet *Crepidula fornicata*. Reefs of *Sabellaria alveolata* in the bay of Mont Saint Michel, France are becoming increasingly colonized by the Pacific oyster *Magallana gigas* (Dubois *et al.* 2006). Given the high filtration rates of *Magallana gigas*, it is believed that they can out-compete *Sabellaria alveolata* for feeding resources (Dubois *et al.*, 2006). In the Wadden Sea, *Magallana gigas* has replaced blue mussels (Foster-Smith, 2000) suggesting that *Magallana gigas* may impact filter-feeding, reef-forming organisms in general. The reasons underlying the species shift from *Mytilus edulis* to *Magallana gigas* have not been elucidated, however, and may be due to recent changes in climatic conditions (Thieltges, 2005) rather than competitive interactions. It should be noted that even though *Magallana gigas* is distributed throughout UK waters following an initial introduction in 1926 (Linke, 1951) there is currently no evidence, in the absence of any targeted studies, that this species is impacting native *Sabellaria spinulosa* or *Sabellaria alveolata* reefs (Hendrick, *et al.* 2011).

When the slipper limpet *Crepidula fornicata* settles in an area it can increase the amount of pseudofaeces and subsequently the substratum may be altered from hard substratum to soft sediment which will reduce the substratum available for settlement by other species. This was observed when a 28-30% mortality of *Mytilus edulis* occurred after the introduction of *Crepidula fornicata* to mussel beds (Thieltges, 2005). *Crepidula fornicata* has been recorded in association with *Sabellaria spinulosa* reefs at Hastings Shingle Bank (up to 66 individuals per grab, Pearce *et al.*, 2007) and in lower numbers in the East Coast REC area (maximum 4 per grab (Pearce *et al.*, 2011a)). The relationship between *Crepidula fornicata* and *Sabellaria spinulosa* has not been investigated. However, potential impacts on *Sabellaria spinulosa* reefs and the associated epifauna could occur through changes to substratum suitability or other interactions.

However, *Undaria pinnatifida* has received a large amount of research attention as a major Invasive Non-Indigenous Species (INIS) which could out-compete native UK kelp habitats (see Farrell & Fletcher, 2006; Thompson & Schiel, 2012, Brodie *et al.*, 2014; Hieser *et al.*, 2014). *Undaria pinnatifida* was first recorded in Plymouth Sound, UK in 2003 (NBN, 2015) subsequent surveys in 2011 have reported that *Undaria pinnatifida* is widespread throughout Plymouth Sound, colonizing rocky reef habitats. Where *Undaria pinnatifida* is present there was a significant decrease in the abundance of other *Laminaria* species, including *Laminaria hyperborea* (Heiser *et al.*, 2014). In New Zealand, Thompson & Schiel (2012) observed that native fucoids could out-compete *Undaria*

*pinnatifida* and re-dominate the substratum. However, Thompson & Schiel (2012) suggested the furoid recovery was partially due to an annual *Undaria pinnatifida* die back, which as noted by Heiser *et al.* (2014) did not occur in Plymouth Sound, UK. It is unknown whether *Undaria pinnatifida* will out-compete native macroalgae in the UK. However, from 2003-2011 *Undaria pinnatifida* had spread throughout Plymouth Sound, UK, becoming a visually dominant species at some locations within summer months (Hieser *et al.*, 2014). While *Undaria pinnatifida* may replace *Laminaria hyperborea* in some locations within the UK, at the time of writing there is limited evidence available to assess what ecological impacts this invasion may have on *Laminaria hyperborea* associated communities e.g. red seaweeds.

*Undaria pinnatifida* was successfully eradicated on a sunken ship in Clatham Islands, New Zealand, by applying a heat treatment of 70°C (Wotton *et al.*, 2004). However, numerous other eradication attempts have failed, and as noted by Farrell & Fletcher (2006) once established *Undaria pinnatifida* resists most attempts of long-term removal. The biotope is unlikely to fully recover until *Undaria pinnatifida* is fully removed from the habitat, which, is unlikely.

**Sensitivity assessment.** No evidence was found that non-indigenous species are currently significantly impacting *Sabellaria spinulosa* reef biotopes but *Crepidula fornicata* and *Magallana gigas* may pose a potential threat in terms of competition for food and space and so this assessment may require updating in the future as the distributions and interactions between these species are better understood. However, *Undaria pinnatifida* has the potential to out-compete *Laminaria hyperborea* and change the character (and classification) of the biotope. Therefore, resistance is assessed as 'Low' and resilience as 'Very Low' so that the sensitivity of this biotope to the introduction of INIS is assessed as 'High'.

#### Introduction of microbial pathogens

**Medium**

Q: Low A: NR C: NR

**High**

Q: Medium A: Medium C: Medium

**Low**

Q: Low A: Low C: Low

Galls on the blade of *Laminaria hyperborea* and spot disease are associated with the endophyte *Streblonema* sp. although the causal agent is unknown (bacteria, virus or endophyte). The resultant damage to the blade and stipe may increase losses in storms. The endophyte inhibits spore production and, therefore, recruitment and recoverability (Lein *et al.*, 1991). However, no evidence was found for adverse impacts of microbial pathogens on *Sabellaria spinulosa*. Therefore, resistance to the pressure is assessed as 'Medium', based on the possible effects on *Laminaria hyperborea*. However, resilience is probably 'High' so that sensitivity is assessed as 'Low'.

#### Removal of target species

**None**

Q: High A: High C: Medium

**Medium**

Q: Medium A: Medium C: Medium

**Medium**

Q: Medium A: Medium C: Medium

Christie *et al.* (1998) observed *Laminaria hyperborea* habitat regeneration following commercial *Laminaria hyperborea* trawling in south Norway. Within the study area trawling removed all large canopy-forming adult *Laminaria hyperborea*, however, sub-canopy recruits were unaffected. Within 2-3 years of harvesting, a new canopy had formed one metre off the seabed. The associated holdfast communities recovered in six years, however, the epiphytic stipe community did not fully recover within the same period. Christie *et al.* (1998) suggested that kelp habitats were relatively resistant to direct disturbance of *Laminaria hyperborea* canopy.

Recurrent disturbance occurring at a smaller time scale than the recovery period of 2-6 years (stated above) could extend recovery time. Kain (1975) cleared sublittoral blocks of *Laminaria*



*hyperborea* at different times of the year for several years. The first colonizers and succession community differed between blocks and the time of year the blocks were cleared. However, within 2 years of clearance, the blocks were dominated by *Laminaria hyperborea*. Leinaas & Christie (1996) observed *Laminaria hyperborea* re-colonization of "urchin barrens", following removal of urchins. The substratum was initially colonized by filamentous macroalgae and *Saccharina latissima* but *Laminaria hyperborea* dominated the community after 2-4 years. Following disturbance or in areas where recurrent rapid disturbance occurs *Laminaria hyperborea* recruitment could also be affected by interspecific competitive interactions with Invasive Non-Indigenous Species or ephemeral algal species (Brodie *et al.*, 2014; Smale *et al.*, 2013), however, evidence for this is limited and thus not included within this assessment. *Sabellaria spinulosa* has no economic value and is not commercially harvested and itself is not directly impacted by this pressure.

**Sensitivity assessment.** Therefore, resistance is assessed as 'None', and resilience 'Medium' so that the sensitivity of this biotope to the removal of target species is assessed as 'Medium'.

### Removal of non-target species

**Low**

Q: Low A: NR C: NR

**Medium**

Q: Medium A: Medium C: Medium

**Medium**

Q: Low A: Low C: Low

Incidental/accidental removal of *Laminaria hyperborea* from extraction of other marine resources, e.g. fisheries or aggregates, is likely to cause similar effects to that of direct harvesting of *Laminaria hyperborea*. Hence, the same evidence has been used for both pressure assessments. Christie *et al.* (1998) observed *Laminaria hyperborea* habitat regeneration following commercial *Laminaria hyperborea* trawling in south Norway. Within the study area trawling removed all large canopy-forming adult *Laminaria hyperborea*, however, sub-canopy recruits were unaffected. Within 2-3 years of harvesting, a new canopy had formed one metre off the seabed. The associated holdfast communities recovered in six years, however, the epiphytic stipe community did not fully recover within the same period. Christie *et al.* (1998) suggested that kelp habitats were relatively resistant to direct disturbance of *Laminaria hyperborea* canopy.

Recurrent disturbance occurring at a smaller time scale than the recovery period of 2-6 years (stated above) could extend recovery time. Kain (1975) cleared sublittoral blocks of *Laminaria hyperborea* at different times of the year for several years. The first colonizers and succession community differed between blocks and the time of year the blocks were cleared. However, the blocks were dominated by *Laminaria hyperborea* within two years of clearance. Leinaas & Christie (1996) observed *Laminaria hyperborea* re-colonization of 'urchin barrens', following the removal of urchins. The substratum was initially colonized by filamentous macroalgae and *Saccharina latissima* but *Laminaria hyperborea* dominated the community after 2-4 years. Following disturbance or in areas where recurrent rapid disturbance occurs *Laminaria hyperborea* recruitment could also be affected by interspecific competitive interactions with Invasive Non-Indigenous Species or ephemeral algal species (Brodie *et al.*, 2014; Smale *et al.*, 2013), however, evidence for this is limited and thus not included within this assessment.

*Sabellaria spinulosa* biotopes may be removed or damaged by static or mobile gears that target other species. These direct, physical impacts are assessed through the abrasion and penetration of the seabed pressures. Evidence for ecological interactions between *Sabellaria spinulosa* and other species is limited. The removal of *Sabellaria spinulosa* predators as by-catch may be beneficial. *Sabellaria spinulosa* reefs appear to be important nursery areas for commercially targeted flatfish including Dover sole (Bryony Pearce, *pers comm*). Assessment of this indirect effect is limited by the lack of empirical evidence for predator-prey relationships. Stomach

analysis of fish by Pearce (2001) found that juvenile flatfish captured in reef areas including Dover sole, dab and plaice fed preferentially on *Sabellaria spinulosa*. Pearce *et al.* (2011b) found that butterflyfish *Pholis gunnellus* and dragonet *Callionymus lyra* also prey on *Sabellaria spinulosa*. Previous studies have also shown that *Carcinus maenas* feeds on *Sabellaria spinulosa* (Taylor, 1962; Bamber & Irving, 1997). Other invertebrates such as *Pandalus montagui* and *Asterias rubens* found in association with *Sabellaria spinulosa* reefs may also be feeding on the worms or other species associated with the reefs rather than *Sabellaria spinulosa*. Due to the limited information available on predator-prey relationships, the impact of predator removal on *Sabellaria spinulosa* reef biotopes cannot be assessed. Dense aggregations of the brittle star, *Ophiothrix fragilis*, have been suggested to compete with *Sabellaria spinulosa* for space and food and potentially to consume the gametes inhibiting recruitment (George & Warwick 1985). Removal of this species as by-catch could potentially be beneficial to the reef biotopes.

**Sensitivity assessment.** Although the removal of predatory species by commercial fisheries may be beneficial to *Sabellaria spinulosa*, the accidental removal (e.g. as by-catch) of a proportion of the kelp bed and possibly the *Sabellaria spinulosa* crusts could impact the biotope. Therefore, resistance is assessed as '**Low**' and resilience as '**Medium**' so that sensitivity is assessed as '**Medium**'.



## Bibliography

- Achari, G.K., 1974. Polychaetes of the family Sabellariidae with special reference to their intertidal habitat. *Proceedings of the Indian National Science Academy*, **38**, 442-55.
- Bamber, R.N. & Irving, P.W., 1997. The differential growth of *Sabellaria alveolata* (L.) reefs at a power station outfall. *Polychaete Research*, **17**, 9-14.
- Benson, A., Foster-Smith, B., Gubbay, S. & Hendrick, V., 2013. Background document on *Sabellaria spinulosa* reefs. *Biodiversity Series, OSPAR Commission, London*, 25 pp. Available from: <https://www.ospar.org/documents?d=7342>
- Beszczynska-Möller, A., & Dye, S.R., 2013. ICES Report on Ocean Climate 2012. In *ICES Cooperative Research Report*, vol. 321 pp. 73.
- Birkett, D.A., Maggs, C.A., Dring, M.J. & Boaden, P.J.S., 1998b. Infralittoral reef biotopes with kelp species: an overview of dynamic and sensitivity characteristics for conservation management of marine SACs. *Natura 2000 report prepared by Scottish Association of Marine Science (SAMS) for the UK Marine SACs Project.*, Scottish Association for Marine Science. (UK Marine SACs Project, vol VI.), 174 pp. Available from: <http://www.ukmarinesac.org.uk/publications.htm>
- Brodie J., Williamson, C.J., Smale, D.A., Kamenos, N.A., Mieszkowska, N., Santos, R., Cunliffe, M., Steinke, M., Yesson, C. & Anderson, K.M., 2014. The future of the northeast Atlantic benthic flora in a high CO<sub>2</sub> world. *Ecology and Evolution*, **4** (13), 2787-2798. DOI <https://doi.org/10.1002/ece3.1105>
- Brown, K.M. & Richardson, T.D., 1988. Foraging ecology of the southern oyster drill *Thais haemastoma* (Gray): constraints on prey choice. *Journal of Experimental Marine Biology and Ecology*, **114** (2), 123-141.
- Buhs, F., & Reise, K. 1997. Epibenthic fauna dredged from tidal channels in the Wadden Sea of Schleswig-Holstein: spatial patterns and a long-term decline. *Helgoländer Meeresuntersuchungen* **51**: 343-59
- Burrows, M.T., Smale, D., O'Connor, N., Rein, H.V. & Moore, P., 2014. Marine Strategy Framework Directive Indicators for UK Kelp Habitats Part 1: Developing proposals for potential indicators. *Joint Nature Conservation Committee, Peterborough*. Report no. 525.
- Chia, F.S. & Spaulding, J.G., 1972. Development and juvenile growth of the sea anemone *Tealia crassicornis*. *Biological Bulletin, Marine Biological Laboratory, Woods Hole*, **142**, 206-218.
- Christie, H., Fredriksen, S. & Rinde, E., 1998. Regrowth of kelp and colonization of epiphyte and fauna community after kelp trawling at the coast of Norway. *Hydrobiologia*, **375/376**, 49-58.
- Cole, S., Codling, I.D., Parr, W. & Zabel, T., 1999. Guidelines for managing water quality impacts within UK European Marine sites. *Natura 2000 report prepared for the UK Marine SACs Project*. 441 pp., Swindon: Water Research Council on behalf of EN, SNH, CCW, JNCC, SAMS and EHS. [UK Marine SACs Project.], <http://www.ukmarinesac.org.uk/>
- Collins, K., 2005. Dorset marine habitat surveys: maerl, worm reefs, brittlestars, sea fans and seagrass. 2004 field report. *Progress report to English Nature from the School of Ocean and Earth Science*. University of Southampton. [Project Ref: DP1/Dorset/MarineHabitat/04/06]. 14 pp.
- Connor, D.W., Allen, J.H., Golding, N., Howell, K.L., Lieberknecht, L.M., Northen, K.O. & Reker, J.B., 2004. The Marine Habitat Classification for Britain and Ireland. Version 04.05. ISBN 1 861 07561 8. In JNCC (2015), *The Marine Habitat Classification for Britain and Ireland Version 15.03*. [2019-07-24]. Joint Nature Conservation Committee, Peterborough. Available from <https://mhc.jncc.gov.uk/>
- Cooper, K., Boyd, S., Eggleton, J., Limpenny, D., Rees, H. & Vanstaen, K., 2007. Recovery of the seabed following marine aggregate dredging on the Hastings Shingle Bank off the southeast coast of England. *Estuarine, Coastal and Shelf Science*, **75**, 547-58.
- Crisp, D.J. (ed.), 1964. The effects of the severe winter of 1962-63 on marine life in Britain. *Journal of Animal Ecology*, **33**, 165-210.
- Cunningham, P.N., Hawkins, S.J., Jones, H.D. & Burrows, M.T., 1984. The geographical distribution of *Sabellaria alveolata* (L.) in England, Wales and Scotland, with investigations into the community structure of and the effects of trampling on *Sabellaria alveolata* colonies. *Nature Conservancy Council, Peterborough, Contract Report* no. HF3/11/22., University of Manchester, Department of Zoology.
- Davies, A.J., Duineveld, G.C., Lavaleye, M.S., Bergman, M.J., van Haren, H. & Roberts, J.M., 2009. Downwelling and deep-water bottom currents as food supply mechanisms to the cold-water coral *Lophelia pertusa* (Scleractinia) at the Mingulay Reef complex. *Limnology and Oceanography*, **54** (2), 620.
- Dieck, T.I., 1992. North Pacific and North Atlantic digitate *Laminaria* species (Phaeophyta): hybridization experiments and temperature responses. *Phycologia*, **31**, 147-163.
- Dieck, T.I., 1993. Temperature tolerance and survival in darkness of kelp gametophytes (Laminariales: Phaeophyta) - ecological and biogeographical implications. *Marine Ecology Progress Series*, **100**, 253-264.
- Dörjes, J., 1992. Langzeitentwicklung makrobenthischer Tierarten im Jadebussen (Nordsee) während der Jahre 1974 bis 1987. *Senckenbergiana maritima*, **22**, 37-57.
- Dubois, S., Commito, J.A., Olivier, F. & Retière, C., 2006. Effects of epibionts on *Sabellaria alveolata* (L.) biogenic reefs and their associated fauna in the Bay of Mont Saint-Michel. *Estuarine, Coastal and Shelf Science*, **68** (3), 635-646.
- Earll R. & Erwin, D.G. 1983. Sublittoral ecology: the ecology of the shallow sublittoral benthos. Oxford University Press, USA.

- Edyvean, R.G.J. & Ford, H., 1984b. Population biology of the crustose red alga *Lithophyllum incrustans* Phil. 3. The effects of local environmental variables. *Biological Journal of the Linnean Society*, **23**, 365-374.
- Eno, N.C., Clark, R.A. & Sanderson, W.G. (ed.) 1997. *Non-native marine species in British waters: a review and directory*. Peterborough: Joint Nature Conservation Committee.
- Fariñas-Franco, J.M., Allcock, L., Smyth, D. & Roberts, D., 2013. Community convergence and recruitment of keystone species as performance indicators of artificial reefs. *Journal of Sea Research*, **78**, 59-74.
- Fariñas-Franco, J.M., Pearce, B., Porter, J., Harries, D., Mair, J.M. & Sanderson, W.G., 2014. Development and validation of indicators of Good Environmental Status for biogenic reefs formed by *Modiolus modiolus*, *Mytilus edulis* and *Sabellaria spinulosa* under the Marine Strategy Framework Directive. *Joint Nature Conservation Committee*.
- Farrell, P. & Fletcher, R., 2006. An investigation of dispersal of the introduced brown alga *Undaria pinnatifida* (Harvey) Suringar and its competition with some species on the man-made structures of Torquay Marina (Devon, UK). *Journal of Experimental Marine Biology and Ecology*, **334** (2), 236-243.
- Fletcher, R.L., 1996. The occurrence of 'green tides' - a review. In *Marine Benthic Vegetation. Recent changes and the Effects of Eutrophication* (ed. W. Schramm & P.H. Nienhuis). Berlin Heidelberg: Springer-Verlag. [Ecological Studies, vol. 123].
- Foster-Smith, J. (ed.), 2000. *The marine fauna and flora of the Cullercoats District. Marine species records for the North East Coast of England*. Sunderland: Penshaw Press, for the Dove Marine Laboratory, University of Newcastle upon Tyne.
- Foster-Smith, R.L. & Hendrick, V.J., 2003. *Sabellaria spinulosa* reef in The Wash and North Norfolk cSAC and its approaches: Part III, summary of knowledge, recommended monitoring strategies and outstanding research requirements. Rep. 543.
- Fredriksen, S., Sjøtun, K., Lein, T.E. & Rueness, J., 1995. Spore dispersal in *Laminaria hyperborea* (Laminariales, Phaeophyceae). *Sarsia*, **80** (1), 47-53.
- Frieder, C., Nam, S., Martz, T. & Levin, L., 2012. High temporal and spatial variability of dissolved oxygen and pH in a nearshore California kelp forest. *Biogeosciences*, **9** (10), 3917-3930.
- George, C.L. & Warwick, R.M., 1985. Annual macrofauna production in a hard-bottom reef community. *Journal of the Marine Biological Association of the United Kingdom*, **65**, 713-735.
- Gibb, N., Tillin, H.M., Pearce, B. & Tyler-Walters, H., 2014. Assessing the sensitivity of *Sabellaria spinulosa* reef biotopes to pressures associated with marine activities. *Joint Nature Conservation Committee, Peterborough, JNCC report No. 504*, 67 pp. Available from: [http://jncc.defra.gov.uk/PDF/JNCC\\_Report\\_504\\_web.pdf](http://jncc.defra.gov.uk/PDF/JNCC_Report_504_web.pdf)
- Hayward, P.J. & Ryland, J.S. 1998. *Cheilostomatous Bryozoa. Part 1. Aeteoidea - Cribrillinoidea*. Shrewsbury: Field Studies Council. [Synopses of the British Fauna, no. 10. (2nd edition)]
- Heiser, S., Hall-Spencer, J.M. & Hiscock, K., 2014. Assessing the extent of establishment of *Undaria pinnatifida* in Plymouth Sound Special Area of Conservation, UK. *Marine Biodiversity Records*, **7**, e93.
- Hendrick, V., Foster-Smith, R., Davies, A. & Newell, R., 2011. Biogenic Reefs and the Marine Aggregate Industry. *Marine ALSF Science Monograph Series*, Cefas, 60pp.
- Hoare, R. & Hiscock, K., 1974. An ecological survey of the rocky coast adjacent to the effluent of a bromine extraction plant. *Estuarine and Coastal Marine Science*, **2** (4), 329-348.
- Holt, T.J., Jones, D.R., Hawkins, S.J. & Hartnoll, R.G., 1995. The sensitivity of marine communities to man induced change - a scoping report. *Countryside Council for Wales, Bangor, Contract Science Report*, no. 65.
- Holt, T.J., Rees, E.I., Hawkins, S.J. & Seed, R., 1998. Biogenic reefs (Volume IX). An overview of dynamic and sensitivity characteristics for conservation management of marine SACs. *Scottish Association for Marine Science (UK Marine SACs Project)*, 174 pp.
- Irvine, L. M. & Chamberlain, Y. M., 1994. *Seaweeds of the British Isles*, vol. 1. *Rhodophyta*, Part 2B *Corallinales*, *Hildenbrandiales*. London: Her Majesty's Stationery Office.
- JNCC, 2015. The Marine Habitat Classification for Britain and Ireland Version 15.03. (20/05/2015). Available from <https://mhc.jncc.gov.uk/>
- JNCC (Joint Nature Conservation Committee), 1999. *Marine Environment Resource Mapping And Information Database (MERMAID): Marine Nature Conservation Review Survey Database*. [on-line] <http://www.jncc.gov.uk/mermaid>
- Jones, D.J., 1972. Changes in the ecological balance of invertebrate communities in kelp holdfast habitats of some polluted North Sea waters. *Helgolander Wissenschaftliche Meeresuntersuchungen*, **23**, 248-260.
- Jones, L.A., Hiscock, K. & Connor, D.W., 2000. Marine habitat reviews. A summary of ecological requirements and sensitivity characteristics for the conservation and management of marine SACs. *Joint Nature Conservation Committee, Peterborough. (UK Marine SACs Project report.)*. Available from: <http://www.ukmarinesac.org.uk/pdfs/marine-habitats-review.pdf>
- Jones, N.S. & Kain, J.M., 1967. Subtidal algal recolonisation following removal of *Echinus*. *Helgolander Wissenschaftliche Meeresuntersuchungen*, **15**, 460-466.
- Kain, J.M., 1964. Aspects of the biology of *Laminaria hyperborea* III. Survival and growth of gametophytes. *Journal of the Marine Biological Association of the United Kingdom*, **44** (2), 415-433.
- Kain, J.M. & Jones, N.S., 1966. Algal colonisation after removal of *Echinus*. *Proceedings of the International Seaweed Symposium*, **8**, 139-140.
- Kain, J.M., 1967. Populations of *Laminaria hyperborea* at various latitudes. *Helgolander Wissenschaftliche Meeresuntersuchungen*, **15**,

489-499.

Kain, J.M., 1971a. Synopsis of biological data on *Laminaria hyperborea*. FAO Fisheries Synopsis, no. 87.

Kain, J.M., 1971b. The biology of *Laminaria hyperborea* VI Some Norwegian populations. *Journal of the Marine Biological Association of the United Kingdom*, **51**, 387-408.

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

Kain, J.M., 1975b. The biology of *Laminaria hyperborea* VII Reproduction of the sporophyte. *Journal of the Marine Biological Association of the United Kingdom*, **55**, 567-582.

Kain, J.M., 1979. A view of the genus *Laminaria*. *Oceanography and Marine Biology: an Annual Review*, **17**, 101-161.

Kain, J.M., 1987. Photoperiod and temperature as triggers in the seasonality of *Delesseria sanguinea*. *Helgolander Meeresuntersuchungen*, **41**, 355-370.

Kain, J.M., & Norton, T.A., 1990. Marine Ecology. In *Biology of the Red Algae*, (ed. K.M. Cole & Sheath, R.G.). Cambridge: Cambridge University Press.

Karsten, U., 2007. Research note: salinity tolerance of Arctic kelps from Spitsbergen. *Phycological Research*, **55** (4), 257-262.

Kinne, O. (ed.), 1972. *Marine Ecology: A Comprehensive, Integrated Treatise on Life in Oceans and Coastal Waters*, Vol.1, *Environmental Factors*, part 3. New York: John Wiley & Sons.

Kregting, L., Blight, A., Elsässer, B. & Savidge, G., 2013. The influence of water motion on the growth rate of the kelp *Laminaria hyperborea*. *Journal of Experimental Marine Biology and Ecology*, **448**, 337-345.

Lang, C. & Mann, K., 1976. Changes in sea urchin populations after the destruction of kelp beds. *Marine Biology*, **36** (4), 321-326.

Last, K.S., Hendrick V. J, Beveridge C. M & Davies A. J, 2011. Measuring the effects of suspended particulate matter and smothering on the behaviour, growth and survival of key species found in areas associated with aggregate dredging. *Report for the Marine Aggregate Levy Sustainability Fund*,

Lein, T.E., Sjøtun, K. & Wakili, S., 1991. Mass-occurrence of a brown filamentous endophyte in the lamina of the kelp *Laminaria hyperborea* (Gunnerus) Foslie along the southwestern coast of Norway. *Sarsia*, **76** (3), 187-193. DOI <https://doi.org/10.1080/00364827.1991.10413474>

Leinaas, H.P. & Christie, H., 1996. Effects of removing sea urchins (*Strongylocentrotus droebachiensis*): stability of the barren state and succession of kelp forest recovery in the east Atlantic. *Oecologia*, **105**(4), 524-536.

Limpenny D.S., Foster-Smith, R.L., Edwards, T.M., Hendrick, V.J., Diesing, M., Eggleton, J.D., Meadows, W.J., Crutchfield, Z., Pfeifer, S. and Reach, I.S. 2010. Best methods for identifying and evaluating *Sabellaria spinulosa* and cobble reef, Marine Aggregate Levy Sustainability Fund

Linke, O. 1951. Neue Beobachtungen über Sandkorallen-Riffe in der Nordsee. *Natur und Volk* 81: 77-84

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

Maggs, C.A. & Hommersand, M.H., 1993. *Seaweeds of the British Isles: Volume 1 Rhodophycota Part 3A Ceramiales*. London: Natural History Museum, Her Majesty's Stationary Office.

McIntosh, W.C., 1922-1923. *A monograph of the British marine annelids. Vol 4. Part I: Hermellidae - Sabellidae. Part II: Sabellidae - Serpulidae*.

Miller III, H.L., Neale, P.J. & Dunton, K.H., 2009. Biological weighting functions for UV inhibition of photosynthesis in the kelp *Laminaria hyperborea* (Phaeophyceae) 1. *Journal of Phycology*, **45** (3), 571-584.

Mistakidis, M.N., 1951. Quantitative studies of the bottom fauna of Essex oyster grounds. *Fishery Investigations, Series 2*, **17**, 47pp.

Moore, P.G., 1985. Levels of heterogeneity and the amphipod fauna of kelp holdfasts. In *The Ecology of Rocky Coasts: essays presented to J.R. Lewis, D.Sc.* (ed. P.G. Moore & R. Seed), 274-289. London: Hodder & Stoughton Ltd.

Newton, L.C. & McKenzie, J.D., 1995. Echinoderms and oil pollution: a potential stress assay using bacterial symbionts. *Marine Pollution Bulletin*, **31**, 453-456.

Nichols, D., 1981. The Cornish Sea-urchin Fishery. *Cornish Studies*, **9**, 5-18.

Norderhaug, K.M. & Christie, H.C., 2009. Sea urchin grazing and kelp re-vegetation in the NE Atlantic. *Marine Biology Research*, **5** (6), 515-528.

Norton, T.A., 1978. The factors influencing the distribution of *Saccorhiza polyschides* in the region of Lough Ine. *Journal of the Marine Biological Association of the United Kingdom*, **58**, 527-536.

Norton, T.A., 1992. Dispersal by macroalgae. *British Phycological Journal*, **27**, 293-301.

NRA (National Rivers Authority), 1994. *Wash Zone Report*. NRA Huntingdon.

Pearce, B., Fariñas-Franco, J.M., Wilson, C., Pitts, J., deBurgh, A. & Somerfield, P.J., 2014. Repeated mapping of reefs constructed by *Sabellaria spinulosa* Leuckart 1849 at an offshore wind farm site. *Continental Shelf Research*, **83**, 3-13.

Pearce, B., Hill, J.M., Grubb, L., Harper, G., 2011a. Impacts of marine aggregate extraction on adjacent *Sabellaria spinulosa* aggregations and other benthic fauna. Rep. MEPF 08/P39, The Crown Estate. DOI <https://doi.org/10.13140/RG.2.2.29285.91361>

Pearce, B., Hill, J.M., Wilson, C., Griffin, R., Earnshaw, S., Pitts, J. 2011b. *Sabellaria spinulosa* reef ecology and ecosystem services The Crown Estate. DOI: <https://doi.org/10.13140/2.1.4856.0644>

- Pearce, B., Taylor, J., Seiderer, L.J. 2007. Recoverability of *Sabellaria spinulosa* Following Aggregate Extraction: Marine Ecological Surveys Limited.
- Pedersen, M.F., Nejrup, L.B., Fredriksen, S., Christie, H. & Norderhaug, K.M., 2012. Effects of wave exposure on population structure, demography, biomass and productivity of the kelp *Laminaria hyperborea*. *Marine Ecology Progress Series*, **451**, 45-60.
- Penfold, R., Hughson, S., & Boyle, N., 1996. *The potential for a sea urchin fishery in Shetland*. <http://www.nafc.ac.uk/publish/note5/note5.htm>, 2000-04-14
- Philippart, C.J., Anadón, R., Danovaro, R., Dippner, J.W., Drinkwater, K.F., Hawkins, S.J., Oguz, T., O'Sullivan, G. & Reid, P.C., 2011. Impacts of climate change on European marine ecosystems: observations, expectations and indicators. *Journal of Experimental Marine Biology and Ecology*, **400** (1), 52-69.
- Reise, K., Herre, E., & Sturm, M. 1989. Historical changes in the benthos of the Wadden Sea around the island of Sylt in the North Sea. *Helgoländer Meeresuntersuchungen*, **43**, 417-433.
- Reise, R. & Schubert, A., 1987. Macrobenthic turnover in the subtidal Wadden Sea: the Norderaue revisited after 60 years. *Helgoländer Meeresuntersuchungen*, **41**, 69-82.
- Riesen, W. & Reise, K., 1982. Macrobenthos of the subtidal Wadden Sea: revisited after 55 years. *Helgoländer Meeresuntersuchungen*, **35**, 409-423.
- Rinde, E. & Sjøtun, K., 2005. Demographic variation in the kelp *Laminaria hyperborea* along a latitudinal gradient. *Marine Biology*, **146** (6), 1051-1062.
- Roberts, D.A., Johnston, E.L. & Knott, N.A., 2010b. Impacts of desalination plant discharges on the marine environment: A critical review of published studies. *Water Research*, **44** (18), 5117-5128.
- Rostron, D.M. & Bunker, F. St P.D., 1997. An assessment of sublittoral epibenthic communities and species following the Sea Empress oil spill. A report to the Countryside Council for Wales from Marine Seen & Sub-Sea Survey., Countryside Council for Wales, Bangor, CCW Sea Empress Contact Science, no. 177.
- Sjøtun, K. & Fredriksen, S., 1995. Growth allocation in *Laminaria hyperborea* (Laminariales, Phaeophyceae) in relation to age and wave exposure. *Marine Ecology Progress Series*, **126**, 213-222.
- Smale, D.A., Burrows, M.T., Moore, P., O'Connor, N. & Hawkins, S.J., 2013. Threats and knowledge gaps for ecosystem services provided by kelp forests: a northeast Atlantic perspective. *Ecology and evolution*, **3** (11), 4016-4038.
- Smale, D.A., Wernberg, T., Yunnice, A.L.E. & Vance, T., 2015. The rise of *Laminaria ochroleuca* in the Western English Channel (UK) and comparisons with its competitor and assemblage dominant *Laminaria hyperborea*. *Marine Ecology*, **36** (4), 1033-1044. DOI <https://doi.org/10.1111/maec.12199>
- Solé-Cava, A.M., Thorpe, J.P. & Todd, C.D., 1994. High genetic similarity between geographically distant populations in a sea anemone with low dispersal capabilities. *Journal of the Marine Biological Association of the United Kingdom*, **74**, 895-902.
- Staehr, P.A. & Wernberg, T., 2009. Physiological responses of *Ecklonia radiata* (Laminariales) to a latitudinal gradient in ocean temperature. *Journal of Phycology*, **45**, 91-99.
- Steneck, R.S., Graham, M.H., Bourque, B.J., Corbett, D., Erlandson, J.M., Estes, J.A. & Tegner, M.J., 2002. Kelp forest ecosystems: biodiversity, stability, resilience and future. *Environmental conservation*, **29** (04), 436-459.
- Steneck, R.S., Vavrinc, J. & Leland, A.V., 2004. Accelerating trophic-level dysfunction in kelp forest ecosystems of the western North Atlantic. *Ecosystems*, **7** (4), 323-332.
- Taylor, A.M., 1962. Notes on the radioecology of Sellafeld beach. *PG Report 353. UK Atomic Energy Authority Production Group*, 20 pp.
- Taylor, P.M. & Parker, J.G., 1993. *An Environmental Appraisal: The Coast of North Wales and North West England*. , Hamilton Oil Company Ltd.
- Thieltges, D.W., 2005. Impact of an invader: epizootic American slipper limpet *Crepidula fornicata* reduces survival and growth in European mussels. *Marine Ecology Progress Series*, **286**, 13-19.
- Thompson, G.A. & Schiel, D.R., 2012. Resistance and facilitation by native algal communities in the invasion success of *Undaria pinnatifida*. *Marine Ecology, Progress Series*, **468**, 95-105.
- Tillin, H.M., 2010. Marine Ecology: Annex 4 Ecological (logistic regression and HABMAP) modelling based predictions., *Parsons Brinkerhoff Ltd, Bristol*.
- Tillin, H.M., Houghton, A.J., Saunders, J.E., Drabble, R. & Hull, S.C., 2011. Direct and Indirect Impacts of Aggregate Dredging. *Marine ALSF Science Monograph Series*, MEPF 10/P144., 41 pp.
- Vorberg, R., 2000. Effects of shrimp fisheries on reefs of *Sabellaria spinulosa* (Polychaeta). *ICES Journal of Marine Science*, **57**, 1416-1420.
- Walker, A.J.M. & Rees, E.I.S., 1980. Benthic ecology of Dublin Bay in relation to sludge dumping: fauna. *Irish Fisheries Investigation Series B (Marine)*, **22**, 1-59.
- Wilson, D.P., 1970a. Additional observations on larval growth and settlement of *Sabellaria alveolata*. *Journal of the Marine Biological Association of the United Kingdom*, **50**, 1-32.
- Wilson, D.P., 1970b. The larvae of *Sabellaria spinulosa* and their settlement behaviour. *Journal of the Marine Biological Association of the United Kingdom*, **50**, 33-52.

- Wilson, D.P., 1971. *Sabellaria* colonies at Duckpool, North Cornwall 1961 - 1970 *Journal of the Marine Biological Association of the United Kingdom*, **54**, 509-580.
- Wotton, D.M., O'Brien, C., Stuart, M.D. & Fergus, D.J., 2004. Eradication success down under: heat treatment of a sunken trawler to kill the invasive seaweed *Undaria pinnatifida*. *Marine Pollution Bulletin*, **49** (9), 844-849.