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Laminaria hyperborea with dense foliose red seaweeds on exposed infralittoral rock

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

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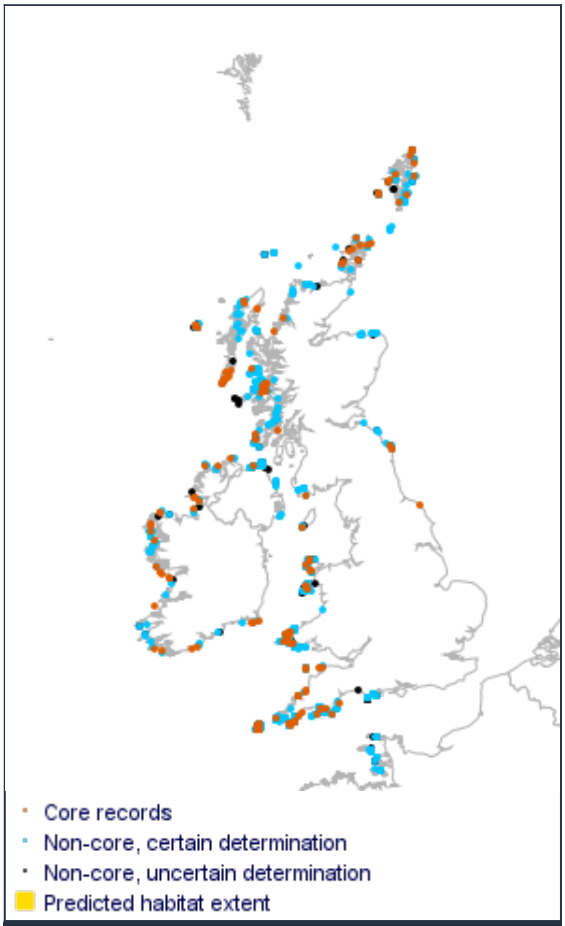
***Laminaria hyperborea* forest with dense foliose red seaweeds on exposed upper infralittoral rock (EIR.LhypR.Ft).**
Photographer: Anon.



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

Researched by Thomas Stamp and Dr Harvey Tyler-Walters **Refereed by** Dr Joanna Jones

Summary

UK and Ireland classification

EUNIS 2008	A3.115	*Laminaria hyperborea* with dense foliose red seaweeds on exposed infralittoral rock
JNCC 2015	IR.HIR.KFaR.LhypR	*Laminaria hyperborea* with dense foliose red seaweeds on exposed infralittoral rock
JNCC 2004	IR.HIR.KFaR.LhypR	*Laminaria hyperborea* with dense foliose red seaweeds on exposed infralittoral rock
1997 Biotope	IR.EIR.KFaR.LhypR	*Laminaria hyperborea* with dense foliose red seaweeds on exposed infralittoral rock

Description

Very exposed and exposed upper infralittoral bedrock or large boulders characterized by the kelp *Laminaria hyperborea*, beneath which is a dense turf of foliose red seaweeds. Three variations of

this biotope have been described: the upper infralittoral kelp forest (EIR.LhypR.Ft), the kelp park below (EIR.LhypR.Pk) and a third type of kelp forest that is characterized by a mixture of *Laminaria hyperborea* and *Laminaria ochroleuca* (EIR.LhypR.Loch). The fauna of EIR.LhypR biotopes are markedly less abundant than kelp forests in areas of greater wave surge (EIR.LhypFa); sponges, anemones and polyclinid ascidians may be present, though never at high abundance. Beneath the understorey of red algae the rock surface is generally covered with encrusting coralline algae. (Information from the Marine Biotope Classification for Britain and Ireland, Version 97.06: Connor *et al.*, 1997a, b).

Depth range

0-5 m, 5-10 m, 10-20 m

Additional information

-

Listed By

- none -

Further information sources

Search on:

G G JNCC

Habitat review

Ecology

Ecological and functional relationships

Kelps are major primary producers, up to 90 percent 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 beds are diverse species rich habitats and over 1,800 species have been recorded in the UK kelp biotopes (Birkett *et al.*, 1998b).

- Sea-urchins graze the undercanopy and understorey 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. Sea urchin grazing may maintain the patchy and species rich understorey epiflora/fauna by preventing dominant species from becoming established. Vost (1983) examined the effect of removing grazing *Echinus esculentus* and found that after 6-10 months the patchiness of the understorey algae had decreased and the species richness and biomass of epilithic species increased. *Strongylocentrotus droebachiensis* and *Paracentrotus lividus* also graze kelp beds but are less common in the British Isles than *Echinus esculentus*. *Echinus esculentus* grazing probably controls the lower limit of *Laminaria hyperborea* distribution in some locations, e.g. in the Isle of Man (Jones & Kain, 1967; Kain *et al.*, 1975; Kain, 1979).
- >Patella pellucida grazes epiphytes and the kelp tissue directly, forming pits similar to the home scars of intertidal limpets (see Kain & Svendsen, 1969 for photographs). The older, *laevis* form excavates large cavities in the holdfast. This tissue damage weakens the adult plant and contributes to its loss due to wave action and storms (Kain, 1979; Birkett *et al.* 1988b). Infestation with *Patella pellucida* varies between sites and decreases with depth, e.g. infestation may reach up to 50 % on mature plants in shallow water in the Isle of Man, whereas <20 % was found (on kelps of any age group and depth) in England and Scotland (Kain, 1979).
- *Laminaria hyperborea*is grazed directly by *Lacuna vincta* in Norway, the individuals forming deep pits in the lamina (Kain, 1979).
- 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 (see Detailed ecology).
- Epiphytes and understorey 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.*, 1988b).
- Predators within kelp beds have not been well studied in the UK. Lobsters (*Homarus gammarus*), crabs and some fish species (e.g. the wolffish *Anarhichas lupus*) are known to consume gastropod and echinoderm grazers. In Scotland, the Eurasian otter *Lutra lutra*, is commonly found feeding on fish in kelp beds (see e.g. Kruuk *et al*, 1990).
- Kelp communities and the interaction between kelp, urchins and predators has been studied in Nova Scotia, Norway, southern California and the UK (Kain, 1979; Mann, 1982; Schiel & Foster, 1986; Elner & Vadas, 1990; Vadas & Elner, 1992; Sivertsen, 1997).
- Birkett *et al.* (1998b) suggest that juveniles of animals present in kelp beds as adults probably use the habitat as a nursery and unknown numbers of species are likely to use the habitat during their life cycle. Rinde *et al.* (1992, cited in Birkett *et al.*, 1998b) state that Norwegian kelp beds are nurseries for gadoid species.

Seasonal and longer term change

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 kelp plants become liable to removal by wave action and storms due to their size and weakening by grazers such as *Patella pellucida*. There is therefore likely to be a reduced abundance of kelps following the winter. Loss of older plants results in more light reaching the understorey, temporarily permitting growth of algae including *Laminaria hyperborea* sporelings. Areas of kelp may become denuded of macroalgae at intervals and the substrata dominated by encrusting corallines. These areas are often associated with an increase in urchin numbers forming 'fronts' of small and large urchins that remove large quantities of algae including the kelps themselves forming 'urchin barrens'. Sea urchin grazing is an important factor in kelp beds and, as part of the biotope, the following suggested factors affecting sea urchin populations are presented.

- Several predators have been suggested as controlling sea urchin populations e.g. sea otters, lobsters, crabs or wolffish, however the evidence is equivocal (Mann, 1982; Elner & Vadas, 1990; Birkett *et al.*, 1998b).
- Evidence suggests that sea urchin recruitment is sporadic and may be enhanced by low temperatures (Birkett *et al.*, 1998b).
- Sea urchin recruitment is also enhanced by the presence of 'urchin barrens' presumably due to the lack of suspension feeders that would otherwise consume their larvae (Lang & Mann, 1978).
- Sea urchin diseases, such as 'bald-urchin' disease, encouraged by high water temperatures drastically reduce the urchin population (Lobban & Harrison, 1997). However, although parasitic infections are found in *Echinus esculentus*, no evidence of sea urchin disease has been found in the UK.
- Sivertsen (1997) examined grazing of west and north Norwegian coast *Laminaria hyperborea* beds by *Strongylocentrotus droebachiensis* and *Echinus esculentus*. He concluded that seven environmental factors contributed to the distribution of kelp beds and 'barrens': depth gradient, latitude, time of sampling, nematode infection (in *Strongylocentrotus droebachiensis*), wave exposure, coastal gradient and substratum.

The factors controlling sea urchin populations and 'urchin barrens' in kelp beds is poorly understood, especially in the UK. However, it is likely that the local urchin population is controlled by a number of factors that vary between sites and biotopes; including predators, competition for food with other grazers, variation in sea urchin recruitment, and parasitic infection or disease.

Periodic storms are likely to remove older and weaker plants creating patches cleared of kelp and increasing the local turbidity. While cleared patches may encourage growth of sporelings or gametophyte maturation, they may also enhance sea urchin recruitment. No studies of storm effects in the UK were found, however, Birkett *et al.* (1998b) cite observations by Dayton *et al.* (1992) of the results of an intense storm (possibly the most severe for 200 years) in a giant kelp forest off Point Loma, San Diego, California. The storm changed the age-specific kelp mortality, caused the first large-scale mortality in the understory and removed drift algae resulting in intense local sea urchin grazing and, therefore, reduced kelp recruitment.

Kelp biotopes should be viewed as dynamic systems that exhibit mixed age kelp stands, together with a patchy distribution of understory flora/fauna and 'urchin barrens'. Kelp communities may exhibit a 15-20 year cycle between 'urchin barrens' and kelp dominated phases (Elner & Vadas, 1990; Lobban & Harrison, 1997) although such clear cycles have not been observed in Britain and

Ireland.

Habitat structure and complexity

Kelp forest provides a variety of habitats and refugia in a similar way to terrestrial forests. Kelps also reduce current flow and their canopy shades the understorey vegetation and substrata producing a particular microclimate depending on the depth and density of the kelp plants. In kelp forest (e.g. EIR.LhypR.Ft) the kelp density produces a canopy which excludes up to 90 % of incident light allowing many deeper water, shade tolerant algae, mainly reds, to invade. In deeper water, as irradiance decreases, the kelp density decreases forming a kelp park. Kelp beds are patchy and dynamic with areas devoid of kelp (due to storms, wave surge or grazing) in the process of expansion or recolonization in different stages of succession. Species diversity changes with depth, between forest & park, with exposure, substratum and turbidity (Norton *et al.*, 1977; Erwin *et al.*, 1990; Birkett *et al.*, 1998b). Erwin *et al.* (1990) noted that species richness increased in the kelp park (as lower infralittoral and upper circalittoral species overlapped) and was higher in boulder fields in which sand-scour and substratum heterogeneity provided more niches for colonization. Kelp beds exhibit a series of stratified habitats, and a patchwork of species depending on the substratum, light, water flow and exposure.

- Planktonic: Spores and larvae from algae and benthic organisms within the bed, as well as from the surrounding area, probably form an import food source given the number of suspension feeding organisms in kelp beds
- Nekton: wrasse and pollock have been observed associated especially with kelp forests and epibenthic predatory or herbivorous fish are also found, e.g. blennies, gobies and wolffish (*Anarhichas lupus*).
- Kelp blades support microalgal epiphytes or endophytes such as *Pogotrichum filiforme*, *Chilionema* sp. and *Myrionema corunnae* which is only found on *Laminaria* blades, as well as *Patella pellucida* and opportunistic hydroids (e.g. *Obelia geniculata*) and bryozoans (e.g. *Membranipora membranacea*).
- The stipes support a diverse fauna and flora, especially foliose red algae (see e.g. Harkin, 1981), depending on age of the stipe, kelp density (stipes in close proximity may abrade each other) and depth. Epiphytes show greater biomass on the top 10-20 cm of stipe and exhibit a zonation pattern down the stipe which changes with depth (Birkett *et al.*, 1998b). Norton *et al.* (1977) found the greatest biomass at 3m depth near Lough Ine. Whittick (1983) showed that epiphyte biomass was significantly greater in plants over 5yrs old, with *Palmaria palmata*(dulse) dominating the top of the stipe from 1-2m, being replaced by *Ptilota plumosa* between 6-10m, while *Membranoptera alata* and *Phycodrys rubens* dominate below 12m or present at lower parts of the stipe. Hiscock & Mitchell (1980) list 15 species of algae associated with kelp stipes in the UK. The stipes also supports epifaunal bryozoa and hydroids (Norton *et al.*, 1977).
- Holdfasts support a diverse fauna that represents a sample of the surrounding mobile fauna and crevice dwelling organisms, e.g., polychaetes, small crabs, gastropods, bivalves, and amphipods. Jones (1971) lists 53 macrofaunal invertebrates in holdfasts and Moore (1973) reports 389 species from holdfasts collected in the north east coast of Britain. A useful account of holdfast fauna is given by Hayward (1988).
- The composition of the holdfast fauna has been shown to vary with turbidity (natural and anthropogenic in origin), between kelp species (due to holdfast architecture and volume), and with location around the coast of the British Isles (Moore, 1973a&b; Moore, 1978; Edwards, 1980; Sheppard *et al.*, 1980). Moore (1973a&b) identified groups of species that were found in most cases, or restricted to either turbid or clear waters. Moore (1978)

noted that the species diversity of amphipods decreased with increasing turbidity, partly due to the increased dominance of a few species. Edwards (1980) noted that holdfast fauna in south-west Ireland were numerically dominated by suspension feeders with decreasing numbers of omnivores and carnivores respectively. Edwards (1980) noted that holdfasts were dominated by *Spirobranchus triqueter* in the most turbid sites, although these were not as turbid as sites examined by Moore (1973 a&b). Sheppard *et al.* (1980) examined 35 sites around the Britain Isles and demonstrated a correlation between heavy metal pollution, turbidity and location. Along the North Sea coast species number and diversity increased with increased clarity, however where heavy metals were a factor species number and diversity decreased with increasing heavy metal pollution. They were able to distinguish groups of species characteristic of all sites, or clear or turbid sties. Along the west coast both heavy metals and turbidity were important. Where turbidity and heavy metals increased suspension feeders increased in abundance while other trophic groups decreased. However, along the south coast longitude was the most important factor, and they suggested that natural variation in temperature, salinity and water flow were responsible for variation between holdfast communities (Sheppard *et al.*, 1980). Moore (1985) also demonstrated that the amphipod fauna varied with water flow rate (resulting from wave action and currents); for example sites of increased exposure were dominated by *Ampithoe rubricata*, *Lembos websteri* and *Jassa falcata* whereas *Gitana sarsi*, *Dexamine thea* and *Corophium bonnellii* flourish in wave sheltered environments.

- A few meiofaunal species may burrow into kelp tissue, e.g. the nematode *Monhystera disjuncta* (Birkett *et al.*, 1998b).
- The understorey flora varies with location, depth, exposure, hydrographic regime, turbidity and siltation and may be sparse or species rich. Birkett *et al.* (1998b; Appendix 5) list 52 common kelp biotope understorey algae in the UK including characterizing species such us *Delesseria sanguinea*, *Dictyota dichotoma*, *Phycodrys rubens*, *Cryptopleura ramosa*, *Plocamium cartilagineum*, and *Callophyllis laciniata*.
- The benthic fauna varies with depth, exposure, location and substratum, however, no species are specific to kelp forest. Norton *et al.* (1977) demonstrate the zonation of 22 epibenthic species. However, many species, both fixed and mobile, are present and probably under recorded (Birkett *et al.*, 1998b).

Productivity

Kelps are the major primary producers in UK marine coastal waters producing nearly 75 percent of the net carbon fixed annually on the shoreline of the coastal euphotic zone (Birkett *et al.*, 1998b). Kelp detritus, as broken plant tissue, particles and dissolved organic material supports soft bottom communities outside the kelp bed itself. As a result, kelp plants can contribute 2-3 times their own biomass to the biomass of the coastal ecosystem over one year (Birkett *et al.*, 1998b). 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.

Recruitment processes

Recruitment processes of key characteristic or dominant species are described here. *Laminaria hyperborea* produces vast numbers of spores, however they need to settle and form gametophytes within about 1 mm of each 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. Recruitment in *Echinus esculentus* is sporadic or annual

depending on location and may benefit from the presence of 'urchin barrens'. *Patella pellucida* is an annual species, larvae settling in the lower eulittoral and juveniles migrating to kelp, via several algal species, as they grow. (View individual key information reviews for details.) Epifaunal larvae probably contribute to the plankton of the kelp bed and many are lost to the suspension feeding epifauna. Kelp beds also provide nurseries for larvae and fish species (see above). Recruitment of epiphytes and epiflora are dependant on dispersal and settlement of algal spores and survival of early post-settlement stages. Norton (1992) suggests that spore dispersal in primarily dependant on currents and eddies. Settlement of algal spores is partly dependant on their motility (if any) and adhesive properties together with preferences for topography (surface roughness), the chemical nature of the substratum and water movement (Norton, 1992; Fletcher & Callow, 1992). Vadas *et al.* (1992) suggested that survival of early post settlement stages is dependant on grazing, the algal canopy and turf effects together with desiccation and water motion, and they further suggest that recruitment is likely to be episodic, variable and to suffer from high mortality of early stages. Kain (1975) examined recolonization of artificially cleared areas in a *Laminaria hyperborea* forest in Port Erin, Isle of Man. Cleared concrete blocks were colonized by *Saccorhiza polyschides*, *Alaria esculenta*, *Desmarestia* spp., *Laminaria hyperborea*, *Laminaria digitata*, *Saccharina latissima* (studied as *Laminaria saccharina*) and un-specified Rhodophyceae at 0.8m. *Saccorhiza polyschides* dominated within 8 months but had virtually disappeared with 77 weeks to be replaced by laminarians, including *Alaria esculenta*. After about 2.5 years, *Laminaria hyperborea* standing crop, together with an understorey of red algae (Rhodophyceae), was similar to that of virgin forest. Rhodophyceae were present throughout the succession increasing from 0.04 to 1.5 percent of the biomass within the first 4 years. Colonizing species varied with time of year, for example blocks cleared in August 1969 were colonized by primarily *Saccharina latissima* and subsequent colonization by *Laminaria hyperborea* and other laminarians was faster than blocks colonized by *Saccorhiza polyschides*; within 1 year the block was occupied by laminarians and Rhodophyceae only. Succession was similar at 4.4m, and *Laminaria hyperborea* dominated within about 3 years. Blocks cleared in August 1969 at 4.4m were not colonized by *Saccorhiza polyschides* but were dominated by Rhodophyceae after 41 weeks, e.g. *Delesseria sanguinea*and *Cryptopleura ramosa*. Kain (1975) cleared one group of blocks at two monthly intervals and noted that Phaeophyceae were dominant colonists in spring, Chlorophyceae (solely *Ulva lactuca*) in summer and Rhodophyceae were most important in autumn and winter. Animal species are likely to recruit mainly from the plankton although some species such as polyclinid tunicates may have only a short lived (2-3 hours) larva (Berrill, 1950). Or no larval stage (amphipods). Little is known about the reproductive biology and dispersal of some species but information from clearance experiments (see 'Time for community to reach maturity) suggests that sponges may be slow to settle.

Time for community to reach maturity

Experimental clearance experiments in the Isle of Man (Kain, 1975; Kain, 1979) showed that *Laminaria hyperborea* out-competed other opportunistic species (e.g. *Alaria esculenta*, *Saccorhiza polyschides* and *Desmarestia* spp.) and returned to near control levels of biomass within 3 years at 0.8 m but that recovery was slower at 4.4m (see above). Studies of the effects of harvesting in Norway (Svendsen, 1972, cited in Birkett *et al.*, 1998b) showed that kelp biomass returned 3-4 years after harvesting, although the plants were small (about 1m) and the age class was shifted towards younger plants. Sivertsen (1991, cited in Birkett *et al.*, 1998b), showed that kelp populations stabilise about 4-5 years after harvesting. Current advice suggests that kelp forest should be left 7-10 years for kelp and non-kelp species to recover (Birkett *et al.*, 1998b). Detailed studies in Norway by Rinde *et al.* (1992, cited in Birkett *et al.*, 1998b) examined recovery of non-kelp species. The epiphyte community in control areas about 10 years old was richer and more extensive than on replacement plants in harvested areas. Of the epifauna, *Halichondria* sp. were

only found on 10 year old plants and tunicates on plants 6 years post harvesting. Holdfast fauna was more abundant richer in 10 year old plants in control areas than younger plants in previously harvested area. Overall his results suggest that full biological recovery, or maturation, may take at least 10 years.

Additional information

No text entered.

Preferences & Distribution

Habitat preferences

Depth Range	0-5 m, 5-10 m, 10-20 m
Water clarity preferences	
Limiting Nutrients	Nitrogen (nitrates), Phosphorus (phosphates)
Salinity preferences	Full (30-40 psu)
Physiographic preferences	Open coast
Biological zone preferences	Infralittoral
Substratum/habitat preferences	Bedrock, Large to very large boulders, Small boulders
Tidal strength preferences	Moderately Strong 1 to 3 knots (0.5-1.5 m/sec.), Weak < 1 knot (<0.5 m/sec.)
Wave exposure preferences	Exposed, Extremely exposed, Very exposed
Other preferences	

Additional Information

Van den Hoek (1982) suggested that the distribution of *Laminaria hyperborea*, and hence its associated biotope, was limited by temperatures between the 2 °C winter isotherm in the north and the 19 °C summer isotherm in the south.

Species composition

Species found especially in this biotope

- *Helcion pellucidum*
- *Membranipora membranacea*
- *Membranoptera alata*
- *Myrionema corunnae*
- *Obelia geniculata*
- *Palmaria palmata*
- *Phycodrys rubens*

Rare or scarce species associated with this biotope

-

Additional information

Little work on the rarity of species in kelp biotopes has been compiled (Birkett *et al.*, 1998b). Kelp beds are diverse species rich habitats and over 1,800 species have been recorded in the UK kelp biotopes (Birkett *et al.*, 1998b). Birkett *et al.* (1998b) list species recorded in UK biotope complexes by the MNCR (Appendix 5) together with common understorey algae and epiphytes (Appendices 4 & 3 respectively).
Holdfast fauna is a particularly species rich part of the biotope but no species have been suggested as specifically associated with holdfasts and therefore critical to the identity of the biotope.

Sensitivity review

Sensitivity characteristics of the habitat and relevant characteristic species

At high densities, *Laminaria hyperborea* forms a canopy over infralittoral rock. Beneath the canopy an understory community grows, typically defined by a red seaweed turf although faunal species dominate in tide swept and/or wave surged conditions. Grazing by the urchins; *Echinus esculentus* and *Paracentrotus lividus* can also define the biotope and reduce the biomass of *Laminaria hyperborea* and understory flora. The abundance of *Laminaria hyperborea* is determined by light availability, which decreases with an increase in water depth. Therefore, depth and water clarity determines the density of *Laminaria* and hence the distribution of kelp forest (high density kelp) and park (low density kelp) sub-biotopes.

Kelp biotopes are a major source of primary productivity, and support magnified secondary productivity within North Atlantic coastal waters (Smale *et al*., 2013, Brodie *et al*., 2014). In Scotland, alone kelp biotopes are estimated to cover 8000 km^2 (Walker, 1953), and account for ca 45% of primary production in UK coastal waters (Smale *et al*., 2013). Therefore kelp biotopes, of which *Laminaria hyperborea* is dominant within UK subtidal rocky reefs (Birkett *et al*., 1998), make a substantial contribution to coastal primary production in the UK (Smale *et al*., 2013). *Laminaria hyperborea* is grazed directly by species such as *Patella pellucida*, however approximately 80% of primary production is consumed as detritus or dissolved organic material (Krumhansl, 2012) which is both retained within and transported out of the parent kelp forest, providing valuable nutrition to potentially low productivity habitats such as sandy beaches (Smale *et al*., 2013).

Laminaria hyperborea also acts as an ecosystem engineer (Jones *et al*., 1994; Smale *et al*., 2013) by altering; light levels (Sjøtun *et al*., 2006), physical disturbance (Connell, 2003), sedimentation rates (Eckman *et al*., 1989) and water flow (Smale *et al*., 2013), profoundly altering the physical environment for fauna and flora in close proximity. *Laminaria hyperborea* biotopes increase the three-dimensional complexity of unvegetated rock (Norderhaug, 2004, Norderhaug *et al*., 2007, Norderhaug & Christie, 2011, Gorman *et al*., 2012; Smale *et al*., 2013) and support high local diversity, abundance and biomass of epi/benthic species (Smale *et al*., 2013), and serve as a nursery ground for a number of commercial important species, e.g. Gadidae (the taxonomic family that contains many commercially important marine fish species, including the Atlantic Cod and Pollack) (Rinde *et al*., 1992).

In undertaking this assessment of sensitivity, an account is taken of knowledge of the biology of all characterizing species/taxa in the biotope. For this sensitivity assessment *Laminaria hyperborea* is the primary focus of research, however, it is recognized that the understory community, typically red seaweeds, also define the biotope. Examples of important species groups are mentioned where appropriate.

Resilience and recovery rates of habitat

A number of 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 6 years (Birkett *et al*., 1998b). Full epiphytic community and stipe habitat complexity regeneration require over 6 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 what time of year the blocks were cleared, however, within two 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 (<25cm) 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 1m within 2-6 years after kelp harvesting. Within 1 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 has a heteromorphic life strategy, A vast number of zoospores (mobile asexual spores) are released into the water column between October-April (Kain & Jones, 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).

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; Steneck *et al.*, 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 (Lienaas & 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; Stenneck *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). While these theories are largely unproven, a number of studies have shown that removal of urchins from grazed areas coincides with kelp re-colonization (Lienaas & Christie, 1996; Nourderhaug & Christie, 2009). Lienaas & 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, Nourderhaug & 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; Stenneck *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 a 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^2 (Lang & Mann, 1976). Jones & Kain (1967) reported urchin abundances of 1-4 /m^2 within experimental plots of the Isle of Man. Therefore, while "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*., 2014). 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*., 2014), 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 sub-tidal rocky reef habitats. The wider ecological consequences of *Laminaria hyperborea'* competition with *Laminaria ochroleuca* and *Undaria pinnatifida* are however as of yet unknown.

Resilience assessment. The evidence suggests that beds of mature *Laminaria hyperborea* can regenerate from disturbance within a period of 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, resilience has therefore been assessed as **Medium.**

Please note, as in Northern Norway, urchin grazing pressure could extend recovery/resilience 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 Invasive 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)

suggested could be a more damaging than the individual pressures.

Hydrological Pressures

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

Kain (1964) stated that *Laminaria hyperborea* sporophyte growth and reproduction could occur within a temperature range of 0 - 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.*, 1988b). Above 17°C gamete survival is reduced (Kain, 1964, 1971a) 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 are also seasonally variable and temperature changes are less tolerated in winter months than summer months (Birkett *et al.*, 1998b).

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). 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.

Laminaria hyperborea has a geographic range from mid-Portugal to Northern Norway (Birkett *et al.*, 1998b), and a mid range within southern Norway (60°-65° North) (Kain, 1971a). The average seawater temperature for southern Norway in October is 12-13°C (Miller *et al.*, 2009), and average annual sea temperature, from 1970-2014, is 8°C (Beszczynska-Möller & Dye, 2013). Against the pressure benchmark, the available information suggests that *Laminaria hyperborea* recruitment processes may be affected and associated red algae communities may decline.

Sensitivity assessment. 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. 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 considered '**Medium**', and resilience '**Medium**'. The sensitivity of this biotope to increases in temperature has been assessed as '**Medium**'.

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

Kain (1964) stated that *Laminaria hyperborea* sporophyte growth and reproduction could occur within a temperature range of 0 - 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.*, 1988b). 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, 1971a). The average seawater temperature for southern Norway in October is 12-13°C (Miller *et al.*, 2009), and average annual sea temperature, from 1970-2014, is 8°C (Beszczynska-Möller & Dye, 2013). The available information suggests that *Laminaria hyperborea* and biotope structure would not be affected by a change in sea temperature at the benchmark level.

Sensitivity assessment. Resistance to the pressure is considered '**High**', and resilience '**High**'. The sensitivity of this biotope to decreases in temperature has been assessed as '**Not Sensitive**'.

Salinity increase (local)

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

Lüning (1990) suggest that kelps are stenohaline, their general tolerance to salinity as a phenotypic group covering 16 - 50 psu over a 24 hr period. Optimal growth probably occurs between 30-35 psu (MNCR category- '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.

Sensitivity assessment. Resistance to the pressure is considered '**Low**', and resilience '**Medium**'. The sensitivity of this biotope to an increase in salinity has been assessed as '**Medium**'.

Salinity decrease (local)

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

Lüning (1990) suggest that kelps are stenohaline, their general tolerance to salinity as a phenotypic group covering 16 - 50 psu over a 24 hr period. Optimal growth probably occurs between 30-35 psu (MNCR category-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, growth was reduced at 16 psu and 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) would 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 as such interspecific interaction between *Laminaria hyperborea* and *Undaria pinnatifida* is not included within this sensitivity assessment.

Sensitivity assessment. Resistance to the pressure is considered '**Low**', and resilience '**Medium**'. The sensitivity of this biotope to decreases in salinity has been assessed as '**Medium**'.

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

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-April 2010. Maximal significant wave height (Hm0) was 3.67 & 2 m at the exposed and sheltered sites, 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. Pederson *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 Pederson *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 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 *et al.*, 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).

Increase water flow rate may also remove or inhibit grazers including *Patella pellucida* and *Echinus esculentus* and remove epiphytic algae growth (Pederson *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).

IR.HIR.KFaR.LhypR, IR.HIR.KFaR.LhypFa, IR.MIR.KR.Lhyp, and their associated sub-biotopes are found within strong (1.5-3 m/s)-moderate (0.5-1.5 m/s) tidal streams. A change in peak mean spring bed flow velocity, which does not result in a change in tidal streams above or below 0.5-3 m/s, is not likely to affect the dominance of *Laminaria hyperborea* within the community, but may cause changes in the understory community. The prominent understory filter feeding community within IR.HIR.KFaR.LhypFa is reliant on high water movement. A decrease in tidal streams may result in a decline of filter feeding fauna and an increase in red seaweeds within the understory community or vice versa with an increase in tidal streams A decrease in tidal flow within this range may also decrease urchin dislodgment and increase urchin grazing. An increase in urchin grazing may cause a decline in the understory community abundance and diversity (as in

IR.MIR.KR.Lhyp.GzFt/Pk and IR.MIR.KR.LhypPar).

Sensitivity assessment. A change in peak mean spring bed flow velocity of between 0.1 m/s to 0.2 m/s for more than 1 year is not likely to affect the dominance of *Laminaria* hyperborea, however, subtle differences in tidal regime may influence the understory community. Resistance to the pressure is considered '**High**', and resilience '**High**'. Hence, the sensitivity of this biotope to changes in peak mean spring bed velocity has been assessed as '**Not Sensitive**'.

However, if peak mean spring bed flow velocity changes but remains within 0.5-3 m/s *Laminaria hyperborea* is likely to remain the dominant habitat but the understory community may be affected; directly by a change in water velocity or through increased grazing pressure.

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

The upper limit of the *Laminaria hyperborea* bed is determined by wave action and 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 water flow), 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 (Birkett *et al.*, 1998b). Hence, an increase in emergence is likely to lead to mortality of exposed *Laminaria hyperborea* and the associated habitat.

An increase in water depth/decreased 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.

Sensitivity assessment. Resistance to the pressure is considered '**Low**', and resilience '**Medium**'. The sensitivity of this biotope to changes in tidal emergence has been assessed as '**Medium**'.

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

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-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 at the exposed and sheltered sites. Maximal water velocity (Velrms) was 0.6 & 0.3 m/s at the exposed and sheltered sites. 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. Pederson *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 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 *et al.*, 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 exposure, *Alaria esculenta* can dominate the shallow sublittoral to a depth of 15 m (Birkett *et al.*, 1998b).

Increase water flow rate may also remove or inhibit grazers including *Patella pellucida* and *Echinus esculentus* and remove epiphytic algae growth (Pederson *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).

IR.HIR.KFaR.LhypR, IR.HIR.KFaR.LhypFa, IR.MIR.KR.Lhyp and their associated sub-biotopes are found between extremely exposed to moderate wave exposure. Changes in local wave height above or below that experienced in extremely exposed to moderately exposed sites will affect the dominance of *Laminaria hyperborea*. Smaller changes in local wave height have the potential to cause changes to the understory community. The prominent understory filter feeding community within IR.HIR.KFaR.LhypFa is reliant on wave surge currents. A decrease in wave surge may result in a decline of filter feeding fauna and an increase in red seaweeds within the understory community or vice versa. A decrease in local wave height may also decrease the chance of urchins being dislodged (removed) from biotopes found at sites with traditionally high wave exposure and may, therefore, increase urchin grazing. An increase in urchin grazing may cause a decline in the understory community abundance and diversity (as in IR.MIR.KR.Lhyp.GzFt/Pk and IR.MIR.KR.LhypPar).

Sensitivity assessment. A change in nearshore significant wave height >3% but <5% is, however, unlikely to have a significant effect. Resistance to the pressure is considered '**High**', and resilience '**High**'. Hence, the sensitivity of this biotope to changes in local wave height has been assessed as '**Not Sensitive**'.

Chemical Pressures

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

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

Bryan (1984) suggested that the general order for heavy metal toxicity in seaweeds is: Organic Hg

> inorganic Hg > Cu > Ag > Zn > Cd > Pb. Cole *et al.*, (1999) reported that Hg was very toxic to macrophytes. Similarly, Hopkin & Kain (1978) demonstrated sub-lethal effects of heavy metals on *Laminaria hyperborea* gametophytes and sporophytes, including reduced growth and respiration. Sheppard *et al.* (1980) noted that increasing levels of heavy metal contamination along the west coast of Britain reduced species number and richness in holdfast fauna, except for suspension feeders which became increasingly dominant. Gastropods may be relatively tolerant of heavy metal pollution (Bryan, 1984). *Echinus esculentus* recruitment is likely to be impaired by heavy metal contamination due to the intolerance of its larvae. *Echinus esculentus* are long-lived and poor recruitment may not reduce grazing pressure in the short-term. Although macroalgae species may not be killed, except by high levels of contamination, reduced growth rates may impair the ability of the biotope to recover from other environmental disturbances.

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

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

Laminaria hyperborea fronds, being almost exclusively subtidal, would not come into contact with freshly released oil, but only to sinking emulsified oil and oil adsorbed onto particles (Birkett *et al.*, 1998b). The mucilaginous slime layer coating of laminarians may protect them from smothering by oil. Hydrocarbons in solution reduce photosynthesis and may be algicidal. However, Holt *et al.* (1995) reported that oil spills in the USA and from the '*Torrey Canyon*' had little effect on kelp forests. Similarly, surveys of subtidal communities at a number sites between 1-22.5m below chart datum, including *Laminaria hyperborea* communities, showed no noticeable impacts of the Sea Empress oil spill and clean up (Rostron & Bunker, 1997). An assessment of holdfast fauna in *Laminaria* showed that although species richness and diversity decreased with increasing proximity to the Sea Empress oil spill, overall the holdfasts contained a reasonably rich and diverse fauna, even though oil was present in most samples (Sommerfield & Warwick, 1999). Laboratory studies of the effects of oil and dispersants on several red algae species, including *Delesseria sanguinea* (Grandy 1984; cited in Holt *et al.*, 1995) concluded that they were all sensitive to oil/ dispersant mixtures, with little differences between adults, sporelings, diploid or haploid life stages. Holt *et al.* (1995) concluded that *Delesseria sanguinea* is probably generally sensitive to chemical contamination. Overall, the red algae are likely to be highly intolerant to hydrocarbon contamination. Loss of red algae is likely to reduce the species richness and diversity of the biotope and the understorey may become dominated by encrusting corallines; however, red algae are likely to recover relatively quickly.

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

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

O'Brian & Dixon (1976) suggested that red algae were the most sensitive group of macrophytes to oil and dispersant contamination (see Smith, 1968). Although *Laminaria hyperborea* sporelings and gametophytes are intolerant of atrazine (and probably other herbicides) overall they may be relatively tolerant of synthetic chemicals (Holt *et al.*, 1995). *Laminaria hyperborea* survived within >55 m from the acidified halogenated effluent discharge polluting Amlwch Bay, Anglesey, albeit at low density. These specimens were greater than 5 years of age, suggesting that spores and/or early stages were more intolerant (Hoare & Hiscock, 1974). *Patella pellucida* was excluded from Amlwch

Bay by the pollution and the species richness of the holdfast fauna decreased with proximity to the effluent discharge; amphipods were particularly intolerant although polychaetes were the least affected (Hoare & Hiscock, 1974). The richness of epifauna/flora decreased near the source of the effluent and epiphytes were absent from *Laminaria hyperborea* stipes within Amlwch Bay. The red alga *Phyllophora membranifolia* was also tolerant of the effluent in Amlwch Bay. Smith (1968) also noted that epiphytic and benthic red algae were intolerant of dispersant or oil contamination due to the Torrey Canyon oil spill; only the epiphytes *Crytopleura ramosa* and *Spermothamnion repens* and some tufts of *Jania rubens* survived together with *Osmundea pinnatifida*, *Gigartina pistillata* and *Phyllophora crispa* from the sublittoral fringe. *Delesseria sanguinea* was probably to most intolerant since it was damaged at depths of 6m (Smith, 1968). Holt *et al.* (1995) suggested that *Delesseria sanguinea* is probably generally sensitive to chemical contamination. Although *Laminaria hyperborea* may be relatively insensitive to synthetic chemical pollution, evidence suggests that grazing gastropods, amphipods and red algae are sensitive. Loss of red algae is likely to reduce the species richness and diversity of the biotope and the understorey may become dominated by encrusting corallines; however, red algae are likely to recover relatively quickly.

Radionuclide contamination	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	No evidence (NEv) Q: NR A: NR C: NR

No evidence was found.

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

This pressure is **Not assessed**.

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

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**).** A 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 2mg/l (Cole *et al.*, 1999).

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 likely to be '**High**', and the biotopes is probably '**Not sensitive**' at the benchmark level.

Nutrient enrichment	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	Not sensitive 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, has 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.

However, this biotope is considered to be '**Not sensitive**' at the pressure benchmark, that assumes compliance with good status as defined by the WFD.

Organic enrichment	Medium Q: Medium A: Medium C: Medium	High Q: High A: Medium C: High	Low 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 sub-littoral untreated sewage outfalls in the Isle of Man. Increased nutrient levels e.g. from sewage outfalls, has 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 are associated with reduced numbers of perennial macrophytes (Fletcher, 1996). Increased nutrients may also result in phytoplankton blooms that increase turbidity. Therefore, although nutrients may not affect kelps directly, indirect effects such as turbidity may significantly affect the structure of *Laminaria hyperborea* biotopes.

Sensitivity assessment. Resistance to the pressure is considered '**Medium**', and resilience '**High**'. The sensitivity of this biotope to organic enrichment is assessed as '**Low**'.

Physical Pressures

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

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

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

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). The biotope would be lost.

Sensitivity assessment. Resistance to the pressure is considered "**None**", and resilience "**Very Low**" or 'None'. The sensitivity of this biotope to change from sedimentary or soft rock substrata to hard rock or artificial substrata or vice-versa 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

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 to hard rock substrata.

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

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 1m off the seabed. The associated holdfast communities recovered in six years, however, the epiphytic stipe community did not fully recover within the same time 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 (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 what time of year the blocks were cleared however within 2 years of clearance the blocks were dominated by *Laminaria hyperborea*. Lienaas & Christie (1996) also observed *Laminaria hyperborea* re-colonization of "urchin barrens", following removal of urchins. The substratum was initially colonized by filamentous macroalgae and *Saccharina latissima* however after 2-4 years *Laminaria hyperborea* dominated the community.

Sensitivity assessment. Resistance to the pressure is considered '**Low**', and resilience '**Medium**'. The sensitivity of this biotope to damage to seabed surface features is assessed as '**Medium**'.

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

Not Relevant; please refer to pressure "Abrasion/disturbance of the substrata on the surface of the seabed".

Changes in suspended solids (water clarity)	None Q: High A: High C: High	Medium Q: High A: High C: High	Medium Q: High A: High C: High

Suspended Particle Matter (SPM) concentration has a linear relationship with sub-surface light attenuation (Kd) (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 in determining *Laminaria hyperborea'* depth range (0-47 m BSL) (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 percent 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 *Laminaria hyperborea* abundance and density (see sub-biotope-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 (Christe *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, and thus, a decrease in water clarity is likely to decrease the abundance of *Laminaria hyperborea* in the affected area (Norton, 1978). An absence of this biotope in silt rich environments is therefore expected.

Sensitivity Assessment. Changes in water clarity are likely to affect photosynthetic rates and enable *Saccharina latissima* to compete more successfully with *Laminaria hyperborea*. A decrease in turbidity is likely to support enhanced growth (and possible habitat expansion) and is therefore not considered in this assessment. An increase in SPM from intermediate to moderate turbidity is likely to significantly reduce the depth at which laminarians can grow. Resistance to this pressure is defined as '**None**' and resilience to this pressure is defined as '**Medium**' at the benchmark level. Hence, this biotope is regarded as having a sensitivity of '**Medium** 'to this pressure.

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

Smothering by sediment e.g. 5 cm material during 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 - 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 1 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 understory epifauna/flora may be adversely affected, e.g.

suspension or filter feeding fauna and/or algal species. This biotope occurs in high wave exposures and, therefore, deposited sediments are unlikely to remain for more than a few tidal cycles, except in the deepest of rock-pools. Therefore, the effects of depositing 5 cm of fine sediment in a discrete event are likely to be transient.

Sensitivity assessment. Resistance to the pressure is considered '**High**', and resilience '**High**'. The sensitivity of this biotope to light deposition of up to 5 cm of fine material added to the seabed in a single discreet event is assessed as '**Note Sensitive**'.

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

Smothering by sediment e.g. 30 cm material during a discrete event is unlikely to damage *Laminaria hyperborea* plants but is likely to affect gametophyte survival, holdfast communities, 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 - 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 understory communities are expected to recover quickly. If inundation is long lasting then the understory epifauna/flora may be adversely affected, e.g. suspension or filter feeding fauna and/or algal species. While this biotope occurs in high to moderate energy habitats (due to water flow or wave action) deposition of 30 cm of sediment represents a large volume of material that would likely remain for a number of tidal cycles and is expected to damage understory flora/fauna as well as juvenile *Laminaria hyperborea*.

Sensitivity assessment. Resistance to the pressure is considered '**Medium**', and resilience '**High**'. The sensitivity of this biotope to heavy deposition of up to 30 cm of fine material added to the seabed in a single discreet event is assessed as '**Low**'.

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

Not assessed.

Electromagnetic changes	Not relevant (NR) 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

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

Not relevant

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

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.

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

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

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

Not relevant. Collision from grounding vessels is addressed under abrasion above.

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

Not relevant

Biological Pressures

	Resistance	Resilience	Sensitivity
Genetic modification & translocation of indigenous species	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	No evidence (NEv) Q: NR A: NR C: NR

No evidence regarding the genetic modification or effects of translocation of native kelp populations was found.

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

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, the UK in 2003 (NBN, 2015) subsequent surveys in 2011 have

reported that *U.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 *U.pinnatifida* and re-dominate the substratum. However, Thompson & Schiel (2012) suggested the fucoid 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 macro-algae 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 (see Wotton *et al.*, 2004) however numerous other eradication attempts have failed, and as noted by Farrell & Fletcher (2006) once established *Undaria pinadifida* resists most attempts at long-term removal. The biotope is unlikely to fully recover until *Undaria pinnatifida* is fully removed from the habitat, which as stated above is unlikely to occur.

Sensitivity assessment. Resistance to the pressure is considered '**Low**', and resilience '**Very Low**'. 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: High A: Low C: High	Low Q: Low A: NR C: NR

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).

Sensitivity assessment. Resistance to the pressure is considered '**Medium**', and resilience '**High**'. The sensitivity of this biotope to the introduction of microbial pathogens is assessed as '**Low**'.

Removal of target species	None Q: High A: High C: High	Medium Q: High A: High C: High	Medium Q: High A: High C: High

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 1 m off the seabed. The associated holdfast communities recovered in 6 years however the epiphytic stipe community did not fully recover within the same time 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 (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 what time of year the blocks were cleared however within two years of clearance the blocks were dominated by *Laminaria hyperborea*. Lienaas & Christie (1996) also observed *Laminaria hyperborea* re-colonization of "urchin barrens", following removal of urchins. The substratum was initially colonized by filamentous macroalgae and *Saccharina latissima* however after 2-4 years *Laminaria hyperborea* dominated the community.

Following disturbance or in areas were recurrent rapid disturbance occurs *Laminaria hyperborea* recruitment could also be affected by interspecific competitive interactions with Invasive Non-Indigenous Species or ephemeral algal species (Smale *et al.*, 2013; Brodie *et al.*, 2014), however, evidence for this is limited and thus not included in this assessment.

Sensitivity assessment. Resistance to the pressure is considered '**None**', and resilience '**Medium**'. The sensitivity of this biotope to damage to seabed surface features is assessed as '**Medium**'.

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

Incidental/accidental removal of *Laminaria hyperborea* from the 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 2-6 years of harvesting, a new canopy had formed 1 m off the seabed. The associated holdfast communities recovered in six years however the epiphytic stipe community did not fully recover within the same time 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 (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 what time of year the blocks were cleared however within two years of clearance the blocks were dominated by *Laminaria hyperborea*. Lienaas & Christie (1996) also observed *Laminaria hyperborea* re-colonization of "urchin barrens", following removal of urchins. The substratum was initially colonized by filamentous macroalgae and *Saccharina latissima* however after 2-4 years *Laminaria hyperborea* dominated the community.

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Sensitivity assessment. Resistance to the pressure is considered '**Low**', and resilience '**Medium**'. The sensitivity of this biotope to damage to seabed surface features is assessed as '**Medium**'.

Bibliography

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 V.). Available from: http://www.ukmarinesac.org.uk/publications.htm

Bishop, G.M., 1985. *Aspects of the reproductive ecology of the sea urchin* Echinus esculentus *L.* Ph.D. thesis, University of Exeter, UK.

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

Bower, S.M., 1996. *Synopsis of Infectious Diseases and Parasites of Commercially Exploited Shellfish: Bald-sea-urchin Disease.* [On-line]. Fisheries and Oceans Canada. [cited 26/01/16]. Available from: http://www.dfo-mpo.gc.ca/science/aah-saa/diseases-maladies/bsudsu-eng.html

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_2 world. *Ecology and Evolution*, **4** (13), 2787-2798.

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

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 Comittee*, Peterborough. Report no. 525.

Casas, G., Scrosati, R. & Piriz, M.L., 2004. The invasive kelp Undaria pinnatifida (Phaeophyceae, Laminariales) reduces native seaweed diversity in Nuevo Gulf (Patagonia, Argentina). *Biological Invasions*, **6** (4), 411-416.

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 [On-line]. *UK Marine SACs Project*. [Cited 26/01/16]. Available from: http://www.ukmarinesac.org.uk/pdfs/water_quality.pdf

Connell, S.D., 2003. The monopolization of understorey habitat by subtidal encrusting coralline algae: a test of the combined effects of canopy-mediated light and sedimentation. *Marine Biology*, **142** (6), 1065-1071.

Connor, D.W., Allen, J.H., Golding, N., Howell, K.L., Lieberknecht, L.M., Northen, K.O. & Reker, J.B., 2004. The Marine Habitat Classification for Britain and Ireland. Version 04.05. ISBN 1 861 07561 8. In JNCC (2015), *The Marine Habitat Classification for Britain and Ireland Version 15.03*. [2019-07-24]. Joint Nature Conservation Committee, Peterborough. Available from https://mhc.jncc.gov.uk/

Connor, D.W., Dalkin, M.J., Hill, T.O., Holt, R.H.F. & Sanderson, W.G., 1997a. Marine biotope classification for Britain and Ireland. Vol. 2. Sublittoral biotopes. *Joint Nature Conservation Committee, Peterborough, JNCC Report* no. 230, Version 97.06., *Joint Nature Conservation Committee, Peterborough, JNCC* Report no. 230, Version 97.06.

Crisp, D.J. (ed.), 1964. The effects of the severe winter of 1962-63 on marine life in Britain. *Journal of Animal Ecology*, **33**, 165-210.

Dauvin, J.C., Bellan, G., Bellan-Santini, D., Castric, A., Francour, P., Gentil, F., Girard, A., Gofas, S., Mahe, C., Noel, P., & Reviers, B. de., 1994. Typologie des ZNIEFF-Mer. Liste des parametres et des biocoenoses des cotes francaises metropolitaines. 2nd ed. *Secretariat Faune-Flore, Museum National d'Histoire Naturelle, Paris (Collection Patrimoines Naturels, Serie Patrimoine Ecologique, No. 12).* Coll. Patrimoines Naturels, vol. 12, Secretariat Faune-Flore, Paris.

Davies, C.E. & Moss, D., 1998. European Union Nature Information System (EUNIS) Habitat Classification. *Report to European Topic Centre on Nature Conservation from the Institute of Terrestrial Ecology, Monks Wood, Cambridgeshire.* [Final draft with further revisions to marine habitats.], Brussels: European Environment Agency.

Dayton, P.K., Tegner, M.J., Parnell, P.E. & Edwards, P.B., 1992. Temporal and spatial patterns of disturbance and recovery in a kelp forest community. *Ecological Monographs*, **62**, 421-445.

Devlin, M.J., Barry, J., Mills, D.K., Gowen, R.J., Foden, J., Sivyer, D. & Tett, P., 2008. Relationships between suspended particulate material, light attenuation and Secchi depth in UK marine waters. *Estuarine, Coastal and Shelf Science*, **79** (3), 429-439.

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.

Dinnel, P.A., Pagano, G.G., & Oshido, P.S., 1988. A sea urchin test system for marine environmental monitoring. In *Echinoderm Biology. Proceedings of the Sixth International Echinoderm Conference, Victoria, 23-28 August 1987*, (R.D. Burke, P.V. Mladenov, P. Lambert, Parsley, R.L. ed.), pp 611-619. Rotterdam: A.A. Balkema.

Eckman, J.E., Duggins, D.O., Sewell, A.T., 1989. Ecology of understory kelp environments. I. Effects of kelps on flow and particle

transport near the bottom. *Journal of Experimental Marine Biology and Ecology*, **129**, 173-187.

Edwards, A., 1980. Ecological studies of the kelp *Laminaria hyperborea* and its associated fauna in south-west Ireland. *Ophelia*, **9**, 47-60.

Elner, R.W. & Vadas, R.L., 1990. Inference in ecology: the sea urchin phenomenon in the northwest Atlantic. *American Naturalist*, **136**, 108-125.

Erwin, D.G., Picton, B.E., Connor, D.W., Howson, C.M., Gilleece, P. & Bogues, M.J., 1990. Inshore Marine Life of Northern Ireland. *Report of a survey carried out by the diving team of the Botany and Zoology Department of the Ulster Museum in fulfilment of a contract with Conservation Branch of the Department of the Environment (N.I.).*, Ulster Museum, Belfast: HMSO.

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].

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.

Gommez, J.L.C. & Miguez-Rodriguez, L.J., 1999. Effects of oil pollution on skeleton and tissues of *Echinus esculentus* L. 1758 (Echinodermata, Echinoidea) in a population of A Coruna Bay, Galicia, Spain. In *Echinoderm Research 1998. Proceedings of the Fifth European Conference on Echinoderms, Milan, 7-12 September 1998*, (ed. M.D.C. Carnevali & F. Bonasoro) pp. 439-447. Rotterdam: A.A. Balkema.

Gorman, D., Bajjouk, T., Populus, J., Vasquez, M. & Ehrhold, A., 2013. Modeling kelp forest distribution and biomass along temperate rocky coastlines. *Marine Biology*, **160** (2), 309-325.

Grandy, N., 1984. *The effects of oil and dispersants on subtidal red algae*. Ph.D. Thesis. University of Liverpool.

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

Harkin, E., 1981. Fluctuations in epiphyte biomass following *Laminaria hyperborea* canopy removal. In *Proceedings of the Xth International Seaweed Symposium, Gø teborg, 11-15 August 1980* (ed. T. Levring), pp.303-308. Berlin: Walter de Gruyter.

Hayward, P.J. 1988. *Animals on seaweed*. Richmond, Surrey: Richmond Publishing Co. Ltd. [Naturalists Handbooks 9].

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.

Hiscock, K. & Mitchell, R., 1980. *The Description and Classification of Sublittoral Epibenthic Ecosystems.* In *The Shore Environment*, Vol. 2, *Ecosystems*, (ed. J.H. Price, D.E.G. Irvine, & W.F. Farnham), 323-370. London and New York: Academic Press. [Systematics Association Special Volume no. 17(b)].

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.

Hopkin, R. & Kain, J.M., 1978. The effects of some pollutants on the survival, growth and respiration of *Laminaria hyperborea*. *Estuarine and Coastal Marine Science*, **7**, 531-553.

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

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

Johansson ,G., Eriksson, B.K., Pedersen, M. & Snoeijs, P., 1998. Long term changes of macroalgal vegetation in the Skagerrak area. *Hydrobiologia*, **385**, 121-138.

Jones, C.G., Lawton, J.H. & Shackak, M., 1994. Organisms as ecosystem engineers. *Oikos*, **69**, 373-386.

Jones, D.J., 1971. Ecological studies on macro-invertebrate communities associated with polluted kelp forest in the North Sea. *Helgolander Wissenschaftliche Meersuntersuchungen*, **22**, 417-431.

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. & Svendsen, P., 1969. A note on the behaviour of *Patina pellucida* in Britain and Norway. *Sarsia*, **38**, 25-30.

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

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

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.

Kain, J.M., Drew, E.A. & Jupp, B.P., 1975. Light and the ecology of *Laminaria hyperborea* II. In *Proceedings of the Sixteenth Symposium of the British Ecological Society, 26-28 March 1974. Light as an Ecological Factor: II* (ed. G.C. Evans, R. Bainbridge & O. Rackham), pp. 63-92. Oxford: Blackwell Scientific Publications.

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

Kinne, O. (ed.), 1984. *Marine Ecology: A Comprehensive, Integrated Treatise on Life in Oceans and Coastal Waters.*Vol. V. *Ocean Management Part 3: Pollution and Protection of the Seas - Radioactive Materials, Heavy Metals and Oil*. Chichester: John Wiley & Sons.

Kinne, O., 1977. International Helgoland Symposium "Ecosystem research": summary, conclusions and closing. *Helgoländer Wissenschaftliche Meeresuntersuchungen*, **30**(1-4), 709-727.

Kitching, J., 1941. Studies in sublittoral ecology III. *Laminaria* forest on the west coast of Scotland; a study of zonation in relation to wave action and illumination. *The Biological Bulletin*, **80** (3), 324-337

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.

Krumhansl, K.A., 2012. *Detrital production in kelp beds*. degree of Doctor of Philosophy, Department of Biology, Dalhousie University, Halifax, Nova Scotia.

Kruuk, H., Wansink, D. & Moorhouse, A., 1990. Feeding patches and diving success of otters, *Lutra lutra*, in Shetland. *Oikos*, **57**, 68-72.

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

Lein, T.E, Sjotun, K. & Wakili, S., 1991. Mass - occurrence of a brown filamentous endophyte in the lamina of the kelp *Laminaria hyperborea* (Gunnerus) Foslie along the south western coast of Norway *Sarsia*, **76**, 187-193.

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.

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

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

Mann, K.H., 1982. Kelp, sea urchins, and predators: a review of strong interactions in rocky subtidal systems of eastern Canada, 1970-1980. *Netherlands Journal of Sea Research*, **16**, 414-423.

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

Moore, P.G., 1973a. The kelp fauna of north east Britain I. Function of the physical environment. *Journal of Experimental Marine Biology and Ecology*, **13**, 97-125.

Moore, P.G., 1973b. The kelp fauna of north east Britain. II. Multivariate classification: turbidity as an ecological factor. *Journal of Experimental Marine Biology and Ecology*, **13**, 127-163.

Moore, P.G., 1978. Turbidity and kelp holdfast Amphipoda. I. Wales and S.W. England. *Journal of Experimental Marine Biology and Ecology*, **32**, 53-96.

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.

NBN, 2015. National Biodiversity Network 2015(20/05/2015). https://data.nbn.org.uk/

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

Norderhaug, K., 2004. Use of red algae as hosts by kelp-associated amphipods. *Marine Biology*, **144** (2), 225-230.

Norderhaug, K.M. & Christie, H., 2011. Secondary production in a *Laminaria hyperborea* kelp forest and variation according to wave exposure. *Estuarine, Coastal and Shelf Science*, **95** (1), 135-144.

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.

Norderhaug, K.M., Christie, H. & Fredriksen, S., 2007. Is habitat size an important factor for faunal abundances on kelp (*Laminaria hyperborea*)? *Journal of Sea Research*, **58** (2), 120-124.

Nordheim, van, H., Andersen, O.N. & Thissen, J., 1996. Red lists of Biotopes, Flora and Fauna of the Trilateral Wadden Sea area,

1995. *Helgolander Meeresuntersuchungen*, **50** (Suppl.), 1-136.

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.

Norton, T.A., Hiscock, K. & Kitching, J.A., 1977. The Ecology of Lough Ine XX. The *Laminaria* forest at Carrigathorna. *Journal of Ecology*, **65**, 919-941.

O'Brien, P.J. & Dixon, P.S., 1976. Effects of oils and oil components on algae: a review. *British Phycological Journal*, **11**, 115-142.

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.

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

Rietema, H., 1993. Ecotypic differences between Baltic and North Sea populations of *Delesseria sanguinea* and *Membranoptera alata*. *Botanica Marina*, **36**, 15-21.

Rinde, E. & Sjøtun, K., 2005. Demographic variation in the kelp *Laminaria hyperborea* along a latitudinal gradient. *Marine Biology*, **146** (6), 1051-1062.

Rinde, E., Christie, H., Fredriksen, S. & Sivertsen, A., 1992. Ecological consequences of kelp trawling: Importance of the structure of the kelp forest for abundance of fauna in the kelp holdfasts, benthic fauna and epiphytes. *Norsk Institutt for Naturforskning. Oppdragsmelding*, (127), 1-37.

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.

Schiel, D.R. & Foster, M.S., 1986. The structure of subtidal algal stands in temperate waters. *Oceanography and Marine Biology: an Annual Review*, **24**, 265-307.

Sheppard, C.R.C. & Bellamy, D.J., 1974. Pollution of the Mediterranean around Naples. *Marine Pollution Bulletin*, **5**, 42-44.

Sheppard, C.R.C., Bellamy, D.J. & Sheppard, A.L.S., 1980. Study of the fauna inhabiting the holdfasts of *Laminaria hyperborea* (Gunn.) Fosl. along some environmental and geographical gradients. *Marine Environmental Research*, **4**, 25-51.

Sivertsen, K., 1997. Geographic and environmental factors affecting the distribution of kelp beds and barren grounds and changes in biota associated with kelp reduction at sites along the Norwegian coast. *Canadian Journal of Fisheries and Aquatic Sciences*, **54**, 2872-2887.

Sjøtun, K., Christie, H. & Helge Fosså, J., 2006. The combined effect of canopy shading and sea urchin grazing on recruitment in kelp forest (*Laminaria hyperborea*). *Marine Biology Research*, **2** (1), 24-32.

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.

Sjøtun, K. & Schoschina, E.V., 2002. Gametophytic development of *Laminaria* spp. (Laminariales, Phaeophyta) at low temperatures. *Phycologia*, **41**, 147-152.

Sjøtun, K., Fredriksen, S. & Rueness, J., 1998. Effect of canopy biomass and wave exposure on growth in *Laminaria hyperborea* (Laminariaceae: Phaeophyta). *European Journal of Phycology*, **33**, 337-343.

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., Yunnie, A.L. & Vance, T., 2014. The rise of *Laminaria ochroleuca* in the Western English Channel (UK) and comparisons with its competitor and assemblage dominant *Laminaria hyperborea*. *Marine ecology*.

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

Somerfield, P.J. & Warwick, R.M., 1999. Appraisal of environmental impact and recovery using *Laminaria* holdfast faunas. *Sea Empress, Environmental Evaluation Committee., Countryside Council for Wales, Bangor, CCW Sea Empress Contract Science, Report* no. 321.

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., Vavrinec, J. & Leland, A.V., 2004. Accelerating trophic-level dysfunction in kelp forest ecosystems of the western North Atlantic. *Ecosystems*, **7** (4), 323-332.

Stock, J.H., 1988. Lamippidae (Copepoda : Siphonostomatoida) parasitic in *Alcyonium*. *Journal of the Marine Biological Association of the United Kingdom*, **68**, 351-359.

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.

Vadas, R.L. & Elner, R.W., 1992. *Plant-animal interactions in the north-west Atlantic.* In *Plant-animal interactions in the marine benthos*, (ed. D.M. John, S.J. Hawkins & J.H. Price), 33-60. Oxford: Clarendon Press. [Systematics Association Special Volume, no. 46].

Vadas, R.L., Johnson, S. & Norton, T.A., 1992. Recruitment and mortality of early post-settlement stages of benthic algae. *British Phycological Journal*, **27**, 331-351.

Van den Hoek, C., 1982. The distribution of benthic marine algae in relation to the temperature regulation of their life histories. *Biological Journal of the Linnean Society*, **18**, 81-144.

Vost, L.M., 1983. The influence of *Echinus esculentus* grazing on subtidal algal communities. *British Phycological Journal*, **18**, 211.

Walker, F. T., 1953. Distribution of Laminariaceae around Scotland. *Journal du Conseil*, **20** (2), 160-166.

Whittick, A., 1983. Spatial and temporal distributions of dominant epiphytes on the stipes of *Laminaria hyperborea* (Gunn.) Fosl. (Phaeophyta: Laminariales) in S.E. Scotland. *Journal of Experimental Marine Biology and Ecology*, **73**, 1-10.

Wilkinson, M., 1995. Information review on the impact of kelp harvesting. *Scottish Natural Heritage Review*, no. 34, 54 pp.

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.