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Marine Strategy Framework Directive Indicators for UK Kelp Habitats
Part 1: Developing proposals for potential indicators

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This project report has been reviewed by a Project Steering Group consisting of experts from the Statutory Nature Conservation Bodies.

Summary

JNCC commissioned a consortium of the Scottish Association for Marine Science (SAMS), the Marine Biological Association (MBA), Queen's University Belfast (QUB) and the University of Aberystwyth (UAber) to deliver an indicator that demonstrates the response to the main pressures acting on UK kelp habitats for the Marine Strategy Framework Directive.

Section 1 provides a review of the information available to support the development of this indicator of Good Environmental Status, and includes a review of the proposed methods for defining and validating the indicator.

Section 2 reports the development of this indicator of Good Environmental Status, and presents evidence underpinning the indicator based on the fitting of species distribution and habitat models for kelp species.

Section 3 identifies major knowledge gaps and gives recommendations about survey designs and protocols that would make the kelp Good Environmental Status indicator operational in the UK.

Knowledge of temporal variability in UK kelp populations is severely lacking, in direct contrast to kelp in other countries, notably the Californian giant kelp. This information is needed to allow the separation of natural (and anthropogenic) climate change and variability from acute and chronic pressures from human activities. Our proposed system of sentinel sites for annual monitoring of kelp should help alleviate this knowledge gap.

Very little is known of the effects of direct human pressures on kelp populations in the UK. We review the evidence available from studies in the north-east of England in the 1970s and conclude that the effects of pollution are hard to disentangle from the effects of turbidity around urban and industrial centres.

Inshore mapping data is also lacking, yet would considerably improve (i) the level of knowledge as to where kelp habitats might be expected to occur and their associated condition, (ii) the predictive ability of models by allowing surface characteristics to be factored in, and (iii) the ability to select survey sites for monitoring work.

The authors recommend a two-tier approach to surveying and monitoring kelp habitats and kelp populations. This would be comprised of (i) a series of sentinel sites surveyed regularly (ideally annually), and (ii) broadscale surveys to assess the state of kelp populations between the survey sites. Details of the likely techniques and protocols are provided.

A straightforward cross-calibration of different techniques for monitoring sublittoral kelp is required, along with an assessment of the efficacy of each technique in order for stronger recommendations to be made. This comparison should address questions of (i) accuracy, (ii) repeatability and (iii) representativeness.

Glossary

ABP	Associated British Ports
BGS	British Geological Survey
DEM	Digital Elevation Model
GEBCO	General Bathymetric Chart of the Oceans
GES	Good Environmental Status - for reporting for MSFD
HSE	Health and Safety Executive
ISR	The Institute of Seaweed Research
MAREMAP	Marine Environmental Mapping Programme
MBA	Marine Biological Association of the United Kingdom
MNCR	Marine Nature Conservation Review
MOB	Man Over Board
MODIS	Moderate Resolution Imaging Spectroradiometer
MSFD	Marine Strategy Framework Directive
NAO	North Atlantic Oscillation
NASA	National Aeronautics and Space Administration
NBN	National Biodiversity Network
NCC	Nature Conservancy Council
NOC	National Oceanography Centre
QUB	Queens University Belfast
ROV	Remotely Operated Vehicle
SAMS	Scottish Association for Marine Science
SST	Sea Surface Temperature

Contents

1 Kelp habitats in the UK – review of available information

1.1 The status of UK kelp research

A recent review by the authors of this report (Smale *et al* 2013) summarises the status of current knowledge and research on kelp in the UK. In contrast to other parts of the world, notably western North America and Australia, kelp has not been a significant focus of coastal ecological research in the UK since the 1970s, even though the habitat was extensively surveyed as part of what became the JNCC's Marine Nature Conservation Review (MNCR) in the 1980s and 1990s. Recent (2013) global efforts to coordinate kelp research and gather data on the vulnerabilities of kelp to pressures from climate change and local anthropogenic impacts, particularly the KEEN network (Kelp Ecosystem Ecology Network: http://www.kelpecosystems.org/), have highlighted this knowledge and effort gap for northern European coastlines. The lack of field-based research currently impedes our ability to conserve and manage kelp in the UK.

The review by Smale *et al* (2013) was not intended to duplicate existing syntheses on the biology and ecology of kelp species (Kain 1979; Dayton 1985), the resilience of kelp beds to perturbation (Steneck *et al* 2002), kelps as drivers of detrital food webs (Krumhansl & Scheibling 2012a) or the likely responses of kelp and other macroalgae to global environmental change (Harley *et al* 2012). Instead, the review aimed to: (i) synthesise existing knowledge on the structure and functioning of kelp beds, and the ecosystem services they provide, in the north-east Atlantic with specific focus on the UK and Ireland; (ii) identify current threats to kelp beds and to assess the likely responses of kelp species and their associated biodiversity to key environmental change stressors; and (iii) highlight pressing knowledge gaps and research priorities for improved understanding of the current and future role of kelp dominated habitats within the wider ecosystem. Within this report, elements of this review have been adapted to support the development of the proposed kelp indicator.

1.1.1 UK kelp research

Quantitative research on UK kelp beds began over 60 years ago, following a demand from the Ministry of Supply to produce camouflage textiles and other goods from kelp-derived alginates during and after the Second World War (Parke 1948; Woodward 1951). In the early 1950s, attempts were made to quantify the total standing stock of kelp as a potentially exploitable resource. The total biomass of sublittoral kelp around Scotland (mostly *Laminaria hyperborea*) was estimated as 10 million tons over an area of 8000km^2 (Walker 1954) (see Section 1.3.1). This figure was a map-based estimate derived from detailed surveys of the coastline (Walker & Richardson 1955) over the period 1946-1955, which included aerial photography and quadrat sampling over an area of 270km^2 (Walker & Richardson 1956). Interestingly, the resultant time series depicted high inter-annual variability in kelp biomass in Scotland which, at the time, was attributed to an 11-year cycle in sunspot activity (Walker 1956b). However, re-examination of the data suggests that the highest annual biomass estimates were recorded in years following North Atlantic Oscillation (NAO) positive summers (Folland *et al* 2009). As such, it could be that calm, sunny weather led to increased biomass, suggesting that decadal and shorter-term NAO variation may be linked to kelp productivity.

Technological advances in scuba diving in the 1960s and 1970s facilitated step-wise progress in our understanding of the distribution and ecology of kelp beds in the UK, particularly through an estimable body of work conducted by Joanna Kain on the ecology of

Laminaria on the Isle of Man (see Kain 1979, for overview), and P.G. Moore's work on faunal assemblages within kelp holdfasts in north-east England (Moore 1971, 1973). Moreover, between 1970 and 2000, substantial survey work was conducted by the Nature Conservancy Council (NCC) and various successor bodies including the Marine Nature Conservation Review (MNCR). During this time, scuba divers conducted semi-quantitative surveys along the majority of the sublittoral rocky coastline of the UK, to benchmark patterns of marine biodiversity. This dataset is freely available and remains the only large-scale, systematic assessment of sublittoral rocky reef assemblages in the UK.

From the 1980s onwards, changes in attitudes and regulations concerning scientific scuba diving, coupled with shifts in research priorities and relatively little commercial interest in kelps, have led to a dearth of primary research on kelp beds in UK waters. Sublittoral kelp beds persist along >12,000 miles of UK coastline yet the volume of directed research in recent years pales in significance when compared with kelp studies conducted in other research-intensive nations: researchers in Australia and the USA published >100 papers on kelps in the last decade, whereas just 7 papers originated from the UK, exceeded by the number from sub-Antarctic regions. The number of papers published on UK rocky shore ecology was 10 times as many as those published on UK kelp habitats. With the notable exception of Norwegian research, kelp ecosystems in the wider north-east Atlantic have been relatively understudied in recent years. Since the structure of kelp beds and current threats off Norway are dissimilar to those further south, generalizing to the wider north-east Atlantic is problematic.

1.1.2 UK kelp habitats

In the UK and Ireland, suitable rocky reef habitat is found along much of the coastline, particularly along the wave-exposed south, west and north coasts. Kelps are found on rocky reefs and artificial hard structures from the low water mark to depths in excess of 40m (e.g. *Alaria esculenta* off Rockall, Scotland). Seven kelp species co-exist, of which four are long-lived climax canopy-forming species; and their relative abundance is influenced by a range of abiotic (e.g. temperature, latitude, wave exposure, light levels, disturbance) and biotic (e.g. competition, grazing) factors. Even so, the dominant canopy-former on most sublittoral reefs is *Laminaria hyperborea*, which is a 'stipitate' kelp species with a rigid stipe (1-3m long) that holds the fronds above the substratum. *L. hyperborea* is distributed from the Arctic south to northern Portugal, and in the UK it persists on all but the most wave-exposed or turbid rocky reefs (Figure 1.1). The sporophyte becomes fertile after 2-6 years and may live for 5-18 years in the UK (Kain 1979). *L. hyperborea* influences its environment and other organisms by providing food and habitat, and by altering light, water motion, sediment deposition and physical disturbance through thallus scour. It is an ecosystem engineer and functions as the assemblage dominant by outcompeting other large macroalgae under most conditions (Hawkins & Harkin 1985).

Other members of the genus found in UK waters are *Laminaria digitata* (Figure 1.1) and *Laminaria ochroleuca. L. digitata* is distributed from Arctic waters to its southern range edge in Brittany, France. It is perennial, reaching maturity after 1-2 years and persisting for up to 6 years, and is smaller than *L. hyperborea*, reaching a maximum total length of 3m. *L. digitata* tends to dominate the low intertidal and immediate sublittoral zones, but is outcompeted by *L. hyperborea* at depths of a few metres (Kain 1975; Hawkins & Harkin 1985). In contrast to *L. hyperborea*, the stipe of *L. digitata* is very flexible so that fronds scour the immediate substratum, which facilitates attachment in the wave-exposed shallow sublittoral zone. *L. ochroleuca* is a warm-temperate Lusitanian species, which is distributed from the south of England to Morocco, and occurs in both the Straits of Messina and the Azores. It is very similar in morphology to *L. hyperborea* and is thought to share similar life history traits, although little is known about its biology in UK waters. *L. ochroleuca* is thought to be expanding its range polewards, perhaps in response to ocean warming. It was first recorded in the far south-west of England and subsequently progressed along the south-west peninsula as far east as the Isle of Wight and northwards onto the north Devon coast (see Blight & Thompson 2008, and references therein). Long-established populations on the south coast are also thought to be increasing in abundance, perhaps at the expense of *L. hyperborea* (Keith Hiscock, pers. comm.).

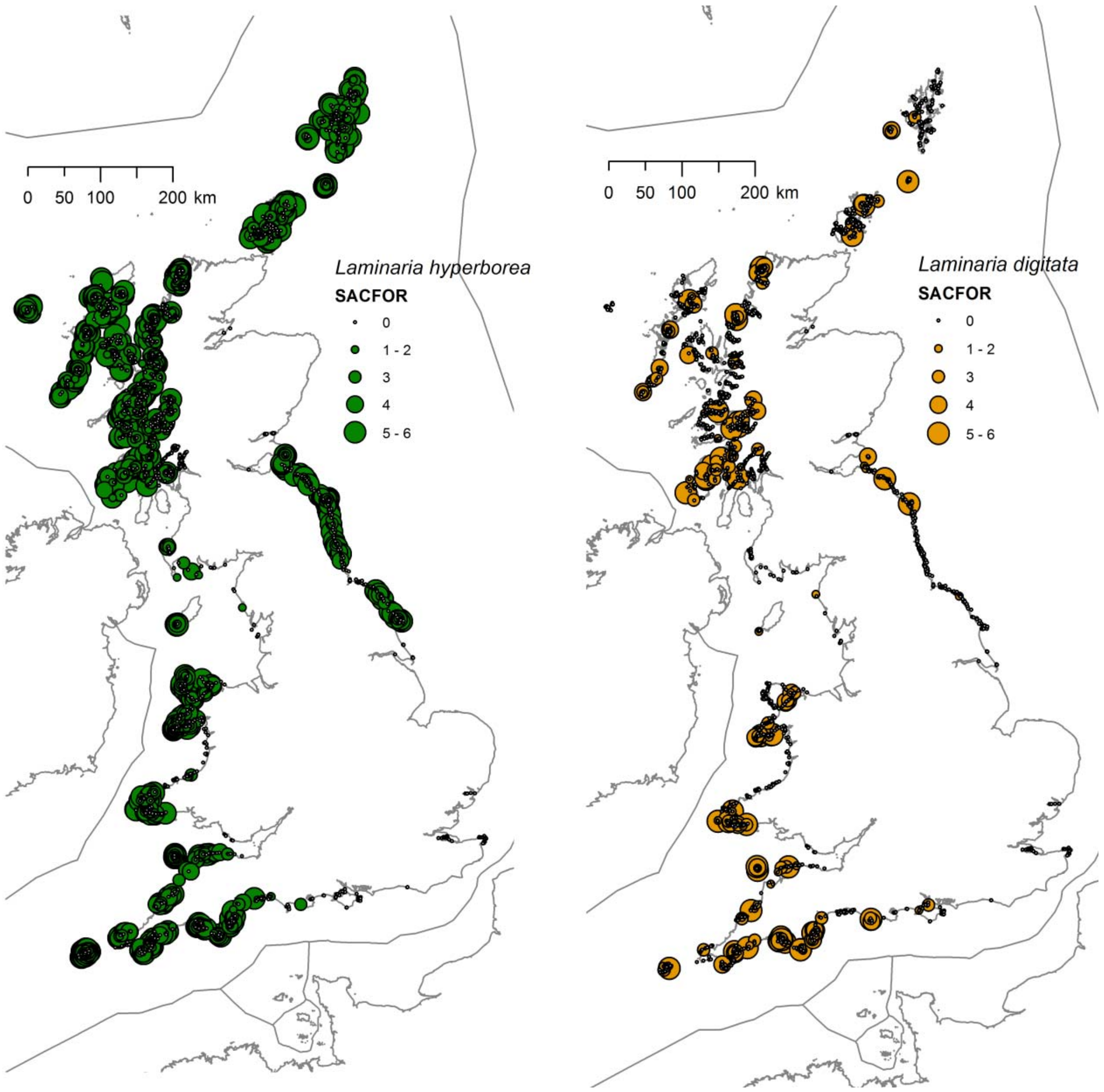

Figure 1.1. The UK distribution of *Laminaria hyperborea* (left) and *Laminaria digitata* (right) as shown by MNCR survey data*.* Abundance categories (SACFOR) are shown as integer values (Super-Abundant, 6; Abundant, 5; Common, 4; Frequent, 3; Occasional, 2; Rare, 1; Absent, 0. See Appendix 1 for definitions).

The remaining kelp species are structurally and functionally diverse and can be locally abundant and sometimes dominant. *Saccharina latissima* (formerly *Laminaria saccharina*) has a short stipe and a single, undivided frond (up to 4m in length) with a 'frilly' undulating margin. It is a short-lived perennial, reaching maturity at 1-2 years and living for up to 4 years. *S. latissima* is found (Figure 1.2) from the Arctic to France (although some isolated populations in northern Portugal may persist) and tends to attach to semi-stable substrata (e.g. boulders) or inhabit the margins of dense *L. hyperborea* beds, particularly in sheltered to moderately exposed locations. *Alaria esculenta* has a similar distribution (Figure 1.2) and, in many respects, morphology (having a short stipe and single blade with distinct midrib extending to 1-3m in length), but is restricted to wave exposed conditions and attaches to stable substrata. It is fertile in about 1 year and lives for 4-7 years. Both species mostly function as mid-successional species and are outcompeted by members of the genus

Laminaria, although under extremely wave-exposed conditions *A. esculenta* may dominate the assemblage (Hawkins & Harkin 1985).

Finally, two short-lived, annual kelp species are found in waters off the UK and Ireland: *Saccorhiza polyschides* and the non-native *Undaria pinnatifida* (‘Wakame’). *S. polyschides* is found from Norway to Morocco, and can be the dominant canopy-former in warmer waters where *L. digitata* and *L. hyperborea* are absent (Hawkins & Harkin 1985). This species is particularly abundant off the south-west coast of Ireland and common throughout much of the UK (Figure 1.3, Norton 1978) . It is a fast-growing opportunistic species that can tolerate very calm through to very turbulent conditions; it attaches to a range of substratum types, and is often found at the margins of dense *Laminaria* beds (Norton 1969). There has been some evidence to suggest that the relative abundance of *S. polyschides* has increased along the south coast of England (Birchenough & Bremmer 2010; Hawkins, pers. comm.), but reliable data are lacking.

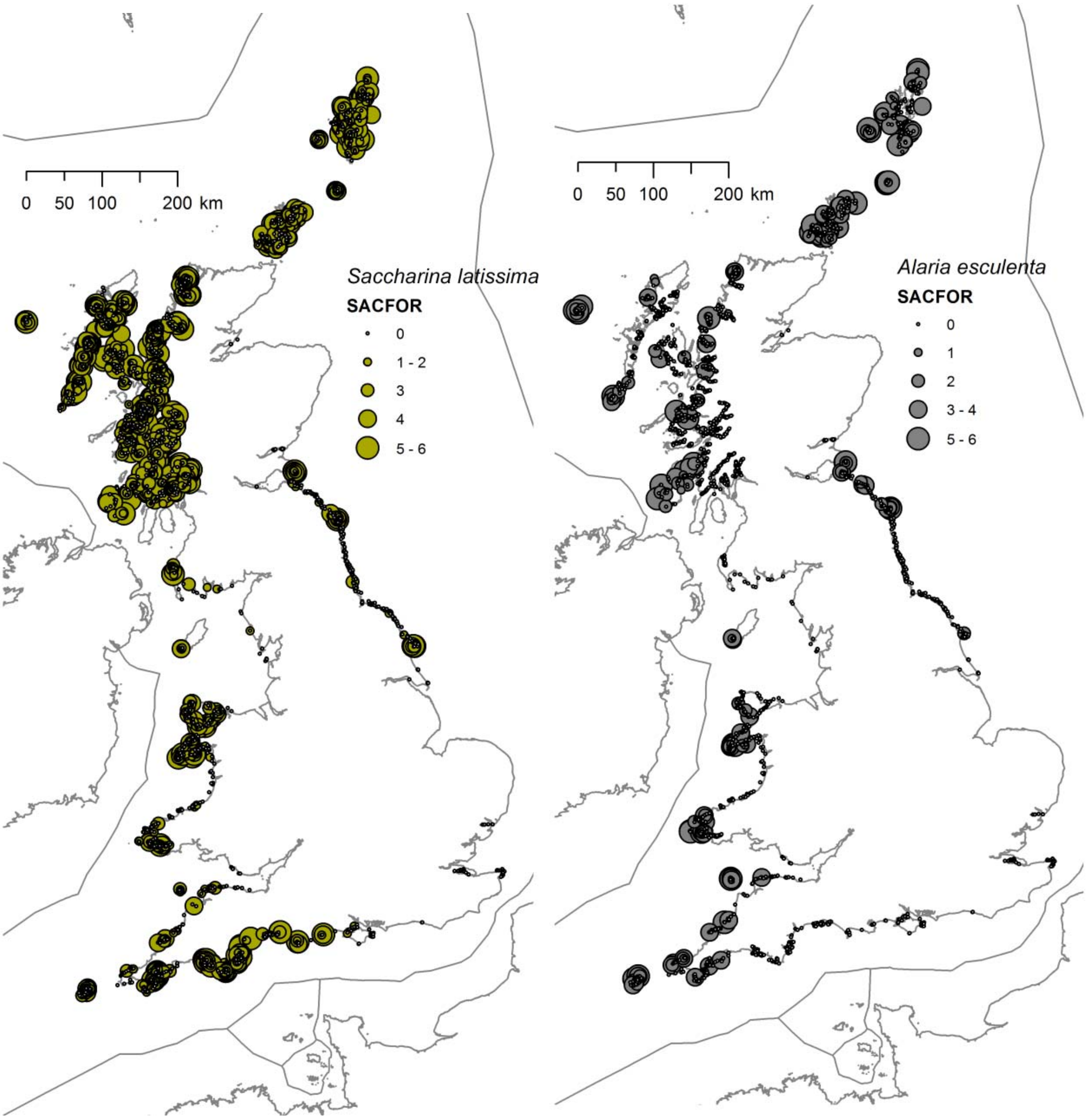

Figure 1.2. The UK distribution of *Saccharina latissima* (left) and *Alaria esculenta* (right). Abundance categories (SACFOR) are shown as integer values (Super-Abundant, 6; Abundant, 5; Common, 4; Frequent, 3; Occasional, 2; Rare, 1; Absent, 0. See Appendix 1 for definitions).

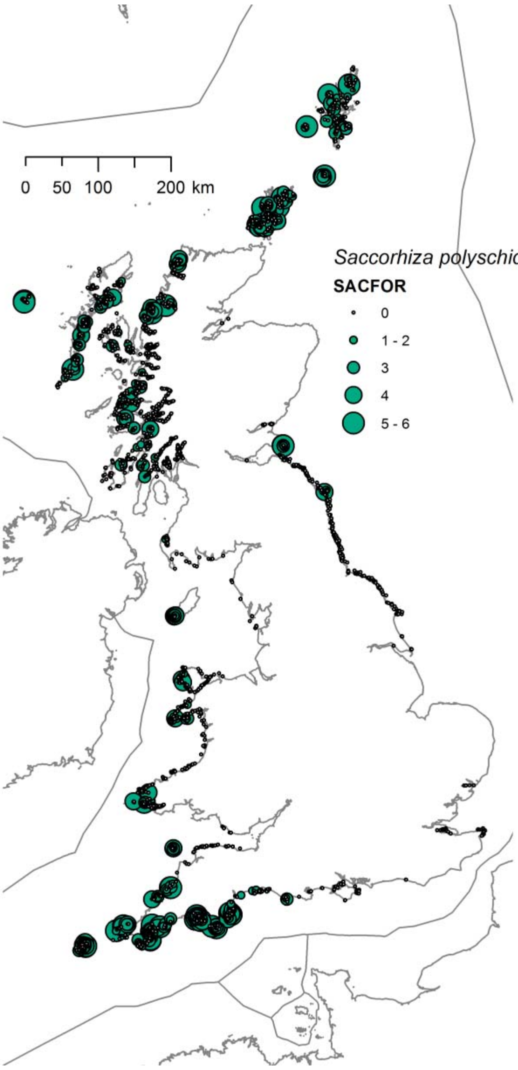

Figure 1.3. The UK distribution of *Saccorhiza polyschides.* Abundance categories are shown as integer values (Superabundant, 6; Abundant, 5; Common, 4; Frequent, 3; Occasional, 2; Rare, 1; Absent, 0. See Appendix 1 for definitions).

There is little doubt, however, that the abundance and distribution of the global invader *Undaria pinnatifida* has increased in UK waters in recent decades; having first been recorded on the south coast of England in 1994 (Fletcher & Manfredi 1995) it has now become established at a number of locations in the UK (Farrell & Fletcher 2006). As it can penetrate low salinity waters, *U. pinnatifida* has become common in some marinas and estuaries. Although *U. pinnatifida* is still restricted to artificial substrates, it is anticipated that this species will colonise natural intertidal and sublittoral habitats.

The structure of entire kelp beds, in terms of the identity and abundance of kelp species and their associated biodiversity, varies considerably in space and time as a function of wave exposure (and storm frequency and magnitude), light levels (influenced by depth and turbidity), sedimentation and temperature. As a general rule, in moderately exposed conditions dense stands of *Laminaria digitata* will persist from the low water mark to a few metres depth (see Figure 2.3), with the upper limit of *Laminaria digitata* set by physical stress and competition with *Fucus serratus* (Hawkins & Harkin 1985) and the lower limit set by competition with *L. hyperborea,* which is mediated by wave exposure. *Saccharina latissima* and *Sacchorhiza polyschides* generally inhabit the immediate sublittoral (see Section 2, Figure 2.1) fringes of rocky reefs or boulders. As the substratum extends into deeper water and light becomes limiting, the density of kelps decreases and isolated (often large) individuals of *L. hyperborea* and *S. polyschides* form ‘park land’. In some locations, such as off the Isle of Man and in Lough Ine, grazing by sea urchins may control the lower depth limit of kelp beds (Kitching & Ebling 1961; Jones & Kain 1967; Kain 1975). While many kelp-dominated systems are dynamic and exhibit pronounced spatio-temporal variability at multiple scales (see Wernberg & Goldberg 2008; Smale *et al* 2010 for Australian examples), others are relatively more stable. For example, southerly distributed European kelp beds (i.e. along the Iberian Peninsula) are more prone to short-term temporal variability arising from variations in both the strength of coastal upwelling and recruitment patterns of dominant canopy formers (e.g. *S. polyschides*). Similarly, high latitude kelp beds may exhibit considerable temporal variability over years to decades, driven by stochastic (or perhaps cyclical) periods of overgrazing by sea urchins, in particular *Strongylocentrotus droebachiensis* (Norderhaug & Christie 2009). It could be that mid-latitude kelp beds are more stable within ecological timescales, although explicit comparisons of variability patterns along broad-scale latitudinal gradients are lacking.

At regional spatial scales across the UK and Ireland, there are some general trends in kelp bed structure that are primarily driven by the abundance distribution patterns of individual kelp species. The occurrence of the cold water kelps *Laminaria hyperborea*, *Saccharina latissima* and *Alaria esculenta* generally increases with latitude from southern England to northern Scotland, moving from the southern limit towards the centre of the species’ ranges. Kelp habitat off the west and north coasts of Scotland is characterised by dense stands of *L. hyperborea* (wave exposed) or *S. latissima* (more sheltered), whereas kelp beds off the south and west coasts of the UK and Ireland are more mixed, with a greater relative abundance of *Sacchorhiza polyschides* and *L. ochroleuca*. This regional-scale shift in kelp bed structure occurs over a latitudinal temperature gradient of approximately 3°C, and may provide some insights into the likely effects of gradual seawater warming on kelp bed structure and function (see Section 1.2.1 on ‘Climate Change’ below).

1.1.3 Kelp as a habitat for other species

Habitat forming species or ‘engineers’ (*sensu* Jones *et al* 1994), such as kelps and corals, exert control over entire communities by modifying the environment and resources available to other organisms (e.g. Bertness & Callaway 1994; Jones *et al* 1997). In particular, kelps alter light, nutrients, sediments, physical scour, and water flow conditions for proximal organisms while providing structural habitat for a wide range of flora and fauna. Within the UK alone, over 1,800 species have been recorded from kelp dominated habitats. As habitat-formers, kelps directly provide three distinct primary habitats; the holdfast, the stipe, and the lamina (Figure 1.4). In addition, epiphytes (primarily attached to the stipe) provide a secondary habitat for colonisation. Over 40 years of descriptive research on kelp-associated faunal assemblages in the north-east Atlantic has shown that kelps host considerable biodiversity (e.g. Moore 1971, 1973; Edwards 1980; Christie *et al* 2003; Blight & Thompson 2008). For example, a study on *Laminaria hyperborea* in Norway by Christie *et al* (2003) showed that, on average, a single kelp plant supports approximately 40 macroinvertebrate species represented by almost 8,000 individuals. The biogenic habitat formed within the kelp holdfast generally harbours the most diverse assemblages, with species richness per holdfast typically in the region of 30-70 macrofaunal species (Edwards 1980; Christie *et al* 2003; Blight & Thompson 2008). However, assemblage richness and structure is strongly influenced by the volume and complexity of the holdfast habitat, as well as external local and regional factors (e.g. turbidity, exposure). The secondary habitat formed by epiphytes on kelp stipes is often used by a highly abundant and diverse fauna (Christie *et al* 2003), which varies considerably in space (i.e. with location and depth) and time (i.e. with season and year). Kelp lamina generally support lower diversity, although epiphyte growth can be very extensive under certain conditions. While diversity may be low, the abundance of several widespread epibionts of kelp lamina (e.g. the blue rayed limpet, *Patella pellucida*, and the 'sea mat' bryozoan *Membranipora membranacea*) can be locally very high (Christie *et al* 2003).

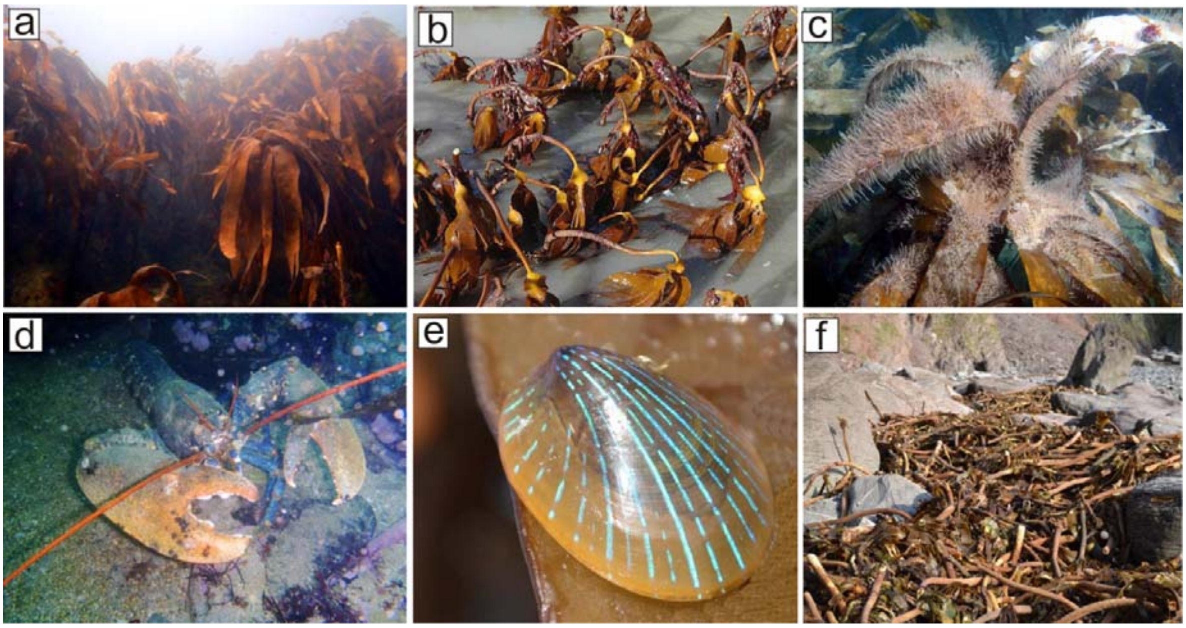

Figure 1.4. The kelp *Laminaria hyperborea* is a dominant canopy-former on both sublittoral (a) and intertidal (b) rocky reefs around the UK and the wider north-east Atlantic. Kelp forests provide habitat for a wide range of flora and fauna, including the hydroid *Obelia geniculata* (c) and the commercially important European Lobster *Homarus gammarus* (d). Although kelps and their epiphytes are grazed directly, by the blue-rayed limpet *Patella pellucida* for example (e), the majority of kelp production is consumed as detritus (f).

At spatial scales larger than that of a single kelp plant, multiple individuals form extensive beds that provide habitat for a vast array of marine organisms. Rich understorey assemblages of plants and animals persist beneath kelp canopies, which ameliorate environmental stressors, and provide shelter and food. With respect to understorey macroalgae, more than 40 species (principally rhodophytes) are regularly found beneath kelp canopies, although their relative abundance varies considerably between biogeographic regions and is strongly influenced by local factors such as depth, turbidity, wave exposure and siltation (Maggs 1986). Studies in other temperate regions have indicated that diverse macroalgal canopies may support greater biodiversity in understory assemblages compared with mono-specific canopy stands (Smale 2010), perhaps because structurally varying canopy-formers enhance habitat diversification. While this has not yet been examined in UK waters, the region represents a tractable model system due to the co-existence of several canopy-forming kelp species.

Kelp beds in the UK and Ireland also provide habitat for large invertebrates, such as gastropod molluscs, crustaceans, and echinoderms, some of which have significant ecological (e.g. sea urchins, see Jones & Kain 1967; Kitching & Thain 1983) or socio-economic (e.g. the European lobster, see Johnson & Hart 2003) importance. Kelp beds are particularly effective nurseries for juvenile invertebrates and fish (e.g. Atlantic cod and pollock), which provide shelter from predation. Moreover, kelp beds are key feeding grounds for many north-east Atlantic fish species, such as *Labrus bergylta* (Ballan wrasse) and *Ctenolabrus rupestris* (Goldsinney wrasse), which prey on kelp-associated invertebrates (Norderhaug *et al* 2005). In turn, elevated fish densities in kelp beds attract large piscivores, such as large fish, seals and otters. In general, sublittoral rocky reefs with extensive stands of *Laminaria hyperborea* support greater species richness than reefs without high kelp coverage (Burrows 2012). Further analysis indicates that species richness on sublittoral rocky reefs around the UK generally increases with increasing relative abundances of all the major canopy-forming kelp species (Figure 1.5).

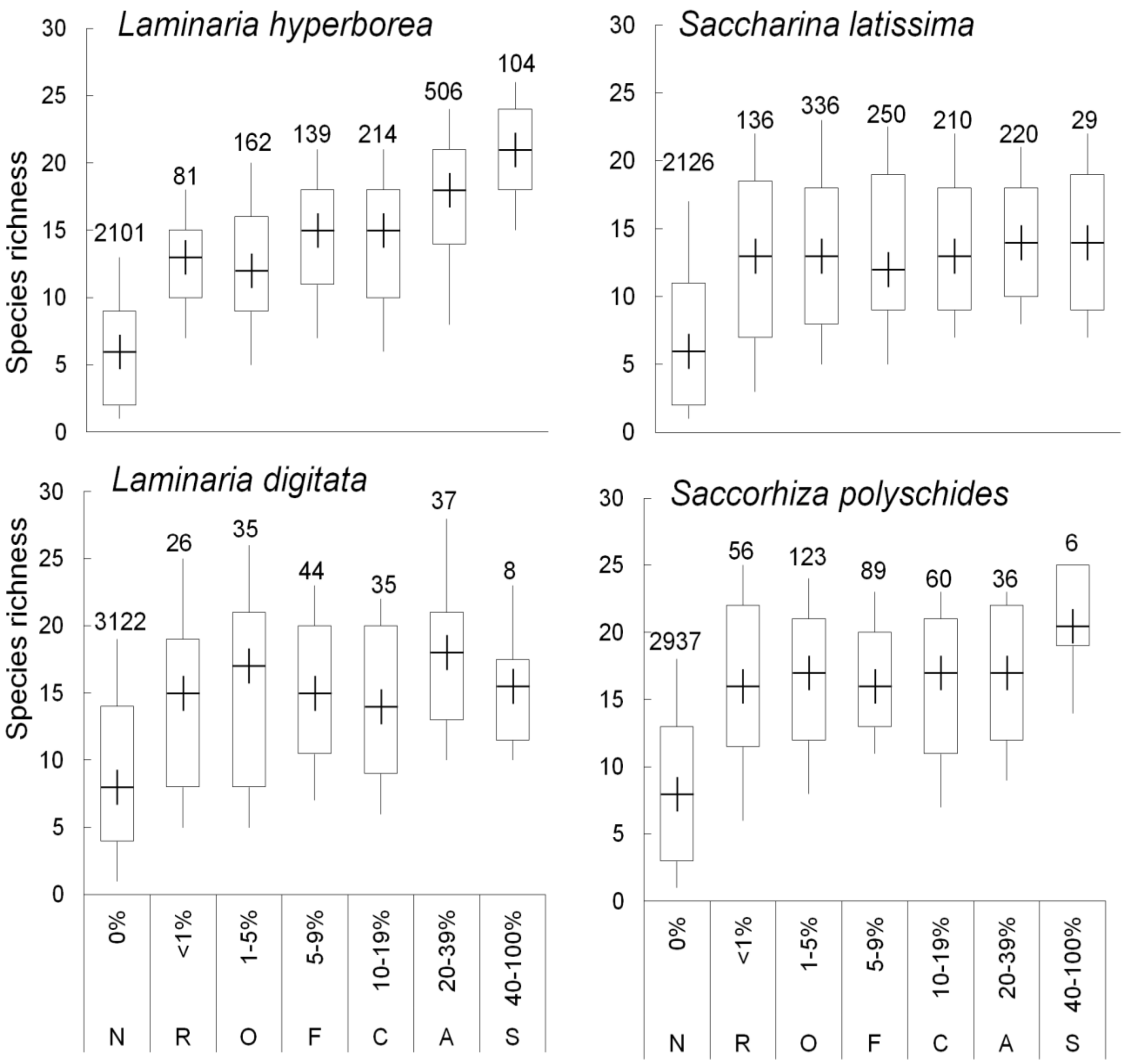

Figure 1.5. Kelp species abundance and local species richness. Box plots show 10th, 25th, 50th, 75th and 90th percentiles of species richness data for each modified SACFOR category of kelp species abundance. For each SACFOR category, *n,* is the number of independent surveys conducted during the Marine Nature Conservation Review (1977-2000).

The vast majority of work on kelps as habitat formers and repositories of biodiversity has focussed on *Laminaria hyperborea*. What is clear, however, is that different kelp species have different morphologies and life histories and, as such, provide structurally varying habitats. This is important within the context of environmental change, because any shifts in the relative abundance of kelp species may have knock-on effects on their associated biodiversity. For example, understorey assemblages associated with *Laminaria digitata* are distinct from those beneath *L. hyperborea* because the stipe of the former is shorter and less rigid. As a result, the substrate near to *L. digitata* plants experiences greater physical abrasion by lamina such that fewer species can inhabit the understorey compared with *L. hyperborea* (Kain 1979). However, certain species such as the limpet *Patella ulyssiponensis* and the sponge *Halichondria panacea* are facilitated by 'sweeping' by *L. digitata,* as they would otherwise be outcompeted by understorey algae. Similarly, subtle differences in morphology (e.g. holdfast volume and complexity, stipe roughness and susceptibility to epiphyte growth) can have a strong influence on the structure and richness of associated assemblages (e.g. Blight & Thompson 2008). The nature of inter-specific and regional-scale variability in kelps as habitat formers within the UK and Ireland (and the wider implications for biodiversity) is poorly understood and remains an important knowledge gap within the field of kelp bed ecology.

1.1.4 Productivity and food webs

Kelp beds represent some of the most productive habitats on Earth (Mann 1973, 2000; Reed *et al* 2008), and are a major source of primary production in coastal zones of temperate and polar oceans worldwide (Steneck *et al* 2002). Kelp productivity is strongly correlated with nutrient availability, but is also affected by temperature, wave exposure, light and disturbance regime (see Reed *et al* 2008, for detailed discussion). Extension (i.e. growth) rates of kelp vary considerably among species and between geographic regions. In southern California maximum growth rates for the giant kelp *Macrocystis pyrifera* can exceed 30 cm per day (Abbott & Hollenberg 1976), whereas members of the genera *Laminaria* and *Saccharina* may exhibit maximum extension rates of ~1cm per day (Parke 1948). Extension rates are closely related to morphology and growth strategy, but when growth rates are converted to biomass production per unit area, different kelp species tend to have broadly comparable productivity rates (Mann 1973; Fairhead & Cheshire 2004; Krumhansl & Scheibling 2012a).

In the Atlantic, kelp primary production can be in excess of 1,000gCm^{-2}yr^{-1}, and that from *Laminaria* species has been estimated at between 110 and 1,780gCm^{-2}yr^{-1} (Mann 1973, 2000), Walker (1954) estimated an area of 2,900km^{-2} of kelp habitat in Scotland alone out of a total sublittoral area of 8,000km^{2}, which may produce 3.6MtCyr^{-1} at typical production rates of 1,300gCm^{-2}yr^{-1} (Dayton 1985). Kelps therefore make a substantial contribution to primary production in coastal waters off the UK and Ireland.

Some kelp biomass is consumed directly by herbivorous fish and invertebrates, such as the conspicuous blue-rayed limpet *Patella pellucida* (Figure 1.4e). However, >80% of kelp production enters the carbon cycle as detritus or dissolved organic matter, since little is directly grazed by herbivores (Krumhansl & Scheibling 2012a). Kelps act as ‘conveyor belts’ of biomass production. At the distal end of the blade, tissue is rapidly or gradually eroded to generate detrital fragments ranging in size from small particulates to large sections of blade. As kelp blades fragment, dissolved organic matter is released, which may account for up to 35% of annual energy production (see Krumhansl & Scheibling 2012a, and references therein). During times of high water motion (i.e. during intense storms or at highly exposed locations), whole kelps may be dislodged following detachment at the holdfast or breakage at the stipe. The proportion of kelp production that is either eroded as fragments or dislodged as whole plants varies amongst species and with morphology and age of kelp. For *Laminaria* spp. and *Saccharina* spp., however, rates of erosion generally exceed rates of dislodgement (Krumhansl & Scheibling 2011).

Kelp detritus is either retained within the kelp bed or exported to adjacent habitats by water movement driven by currents, tides or waves. Rates of export exhibit pronounced spatio-temporal variability as they are governed by a complex, interacting suite of factors including water flow, seabed topography, substratum type, and aspects of the detritus itself (e.g. size, buoyancy, density, age). Kelp detritus may settle locally and form a food source for a wide range of benthic invertebrates (Duggins & Estes 1989; Norderhaug *et al* 2003), or be transported to adjacent (Tallis 2009) or distant habitats (Vanderklift & Wernberg 2008). Either way, most kelp-derived carbon is consumed by suspension feeders, detrital grazers (such as limpets and *Littorina littorea*) and general consumers of organic material in soft sediments (deposit feeders). An important, but poorly understood, process relating to kelp detritus consumption concerns the interactions between microbes and macrofauna. It is clear that microbial degradation of kelp tissue increases palatability for many grazers by reducing carbon:nitrogen ratios and phlorotannin content (Norderhaug *et al* 2003), but the influence of microbial processes on palatability varies between species of kelps (Duggins & Eckman 1997) and grazers (Norderhaug *et al* 2003), and microbial degradation may be less important than for angiosperms such as seagrasses (Bedford & Moore 1984).

Kelp detritus is particularly important as a spatial subsidy of energy into low-productivity habitats. The most visible example being the deposition of kelp wrack into sandy beach habitats, where it provides a principal food source for rich and abundant microbial and faunal assemblages (Ince *et al* 2007). Similarly, exported kelp represents a spatial energy subsidy into seagrass meadows (Wernberg *et al* 2006; Hyndes *et al* 2012), soft sediments (Bedford & Moore 1984; Vetter & Dayton 1998), sublittoral reefs (Vanderklift & Wernberg 2008) and rocky intertidal habitats (Bustamante & Branch 1996; Tallis 2009). Kelp detritus may be consumed many kilometres from its source (Vanderklift & Wernberg, 2008) and, following offshore transportation, may enrich soft sediments at depths of 900m or more (Vetter & Dayton 1998). In the UK and Ireland, targeted research on kelps as fuels of coastal food webs has been lacking, and specific rates of kelp detritus production and export remain almost entirely unknown (but see Johnston *et al* 1977, for experiment on *Saccharina latissima* in Scotland). Evidence from elsewhere would indicate that kelp biomass is a hugely important source of exported energy which influences patterns of secondary production and the distribution of marine organisms. Detritus production and export rates are likely to vary considerably between regions and seasons, and the quantity and quality of exported material will vary between kelp species.

1.2 Threats

1.2.1 Climate change

In Europe, marine plants and animals have undergone climate-driven shifts in their distribution, and major changes in assemblage structure and ecosystem function are projected to occur as a result (Helmuth *et al* 2006; Hawkins *et al* 2009). Ecological responses to recent warming in the north-east Atlantic have included shifts in the distributions of plankton (Pitois and Fox, 2006; Beaugrand *et al* 2009), intertidal invertebrates (Hawkins *et al* 2003; Mieszkowska *et al* 2006) and fish (Genner *et al* 2004), as well as changes in phenology and behaviour (Sims *et al* 2001; Edwards & Richardson 2004; Moore *et al* 2011). For plankton, intertidal invertebrates and fish, access to long-term historical data has facilitated robust examinations of temporal shifts in assemblage structure in response to climate. Patterns of ecological change, and the processes driving them, have been well documented in both intertidal (Helmuth *et al* 2006; Hawkins *et al* 2009) and pelagic (Richardson and Schoeman, 2004) systems, but there is limited information from sublittoral benthic systems, especially from hard-bottom habitats that cannot be routinely trawled, dredged or cored. This was highlighted by the recent ‘Marine Climate Change Impacts Knowledge Gaps’ report, which stated that “*knowledge of large scale benthic species distributions within UK waters is required, to detect changes over large areas of the seabed and patterns of benthic response to climate change*”. This understanding is urgently needed to maintain “*healthy and biologically diverse seas*” (MCCIP 2012).

Kelps are cool-water species that are stressed by high temperatures (Steneck *et al* 2002), so that ocean warming will affect the distribution, structure, productivity and resilience of kelp beds (Dayton *et al* 1992; Wernberg *et al* 2010; Harley *et al* 2012). Poleward range contractions have been predicted for several more northerly-distributed kelp species (e.g. *Alaria esculenta*, *Laminaria digitata*, *Laminaria hyperborea*) in response to ocean warming in the Atlantic (Hiscock *et al* 2004; Muller *et al* 2009). It is evident that the relative abundance of several kelp species changes with latitude along north-east Atlantic coastlines, which corresponds to a regional-scale temperature gradient, and that several habitat-forming kelps are at their range edge in the UK and Ireland, such as *Laminaria ochroleuca* at its northernmost limit and *A. esculenta* at its southernmost limit. Because of these distribution patterns, and because the distributions of some intertidal species have shifted, several authors have predicted that relatively southerly-distributed species will increase in abundance while more northerly-species will decrease in abundance and/or undergo range contractions in the UK and Ireland (Breeman 1990; Hiscock *et al* 2004). It has been suggested that more southerly-distributed kelp species (*L. ochroleuca* and *Saccorhiza polyschides*) have increased in abundance and have undergone poleward range-edge expansions while, conversely, northern species *A. esculenta*) have decreased in abundance in response to recent warming (Simkanin *et al* 2005; Birchenough & Bremmer 2010). However, the evidence to support this is largely anecdotal and based on unpublished survey data; detailed historical examinations of distribution patterns are lacking.

Combined with climate warming, observed and predicted increases in storminess (Lozano *et al* 2004; Weisse *et al* 2005) are likely to have a negative impact on kelp beds, as canopy-forming macroalgae may be damaged and dislodged during periods of intense wave action. Increased storminess and canopy removal will affect the structure and functioning of entire kelp habitats, by altering patch dynamics (Dayton & Tegner 1984) and potentially driving ecological phase shifts (Dayton *et al* 1999; Wernberg *et al* 2011). Crucially, multiple concurrent stressors do not act in isolation but often combine synergistically in their effects, so that the total impact is far greater than the sum of individual factor effects (Crain *et al* 2008; Harvey *et al* 2013). Synergism can cause ‘ecological surprises’, where unexpected regime shifts occur quickly because a tipping point is exceeded (Crain *et al* 2008). In kelp beds, multiple stressors can cause irreversible shifts from complex, biologically diverse habitats to simple turf-dominated ‘barrens’ (Dayton & Tegner 1984; Ling *et al 2*009).
As changes in the identity and abundance of habitat-forming species can have wide-ranging consequences for community structure and ecosystem functioning (Jones *et al* 1994), there is a pressing need to examine climate-driven distribution shifts and their wider implications. For example, if a cool water habitat-former is replaced by a warm water species that is functionally and structurally similar, it is plausible that the wider community or ecosystem will be relatively unaffected (e.g. Terazono *et al* 2012). Conversely, if a structurally or functionally dissimilar species becomes dominant, or habitat formers are lost and not replaced, then widespread changes in biodiversity patterns and ecological processes are likely to ensue (Ling 2008). In the UK and Ireland, a range contraction of *Alaria esculenta*, the dominant species on very exposed shores and an important mid-successional species in more sheltered locations (Hawkins & Harkin 1985), would impact community structure and functioning as there is no warm water equivalent. *A. esculenta* is particularly susceptible to climate fluctuations, having disappeared from much of the English Channel during a warm period in the 1950s, and not recovering as conditions became cooler in the 1960s. Replacement of *Laminaria hyperborea* with *Laminaria ochroleuca*, which are more similar both structurally and functionally, may have fewer knock-on effects, although subtle differences in kelp species traits have been shown to influence local biodiversity patterns (Blight & Thompson 2008). Most dramatically, the predicted increase in the relative abundance of *Saccorhiza polyschides* (Birchenough & Bremmer 2010) could have major implications for kelp bed structure and functioning as it is a fast-growing, annual species with distinct morphological and ecological traits. As kelps make a significant contribution to coastal primary production, facilitate export of carbon from high to low productivity systems, and fuel entire food webs, changes in the quality or quantity of detrital material resulting from climate-driven changes in kelp species identity, abundance or productivity could have far-reaching consequences (Krumhansl & Scheibling 2012a). In the UK and Ireland the wider implications of shifts in kelp species identity and abundance for kelp bed productivity, trophic linkages and ecosystem functioning are almost entirely unknown.

It may be possible to predict the future structure of kelp beds under continued ocean warming in the UK and Ireland by examining the current structure of kelp beds under warmer conditions further south. For example, coastal waters off northern Portugal are approximately 3°C warmer than off southern England and approximately 5°C warmer than north-west Scotland, which is within the projected range of north-east Atlantic warming within the next 50-80 years (Philippart *et al* 2011). The structure of kelp bed habitats off northern Portugal and Spain is strikingly different to those in UK waters (Hawkins & Harkin 1985; Fernandez 2011; Tuya *et al* 2012). Most obviously, the geographical range of *Laminaria digitata* does not extend further south than France and therefore does not form dense stands in the low intertidal and shallow sublittoral zones. *Laminaria hyperborea* is present as far south as northern Portugal, but is generally much smaller and lower in abundance, forming 'parks' rather than dense canopies under warmer conditions. Conversely, *Laminaria ochroleuca* is more abundant and often larger, while *Sacchoriza polyschides* is generally more abundant across a wider depth range. Recent observations indicate that *Sacchoriza polyschides* and probably *Laminaria ochroleuca* (Fernandez 2011) and *L. hyperborea* (Tuya *et al* 2012) have undergone range contractions in recent decades in response to warming off the Iberian Peninsula. In contrast, Lima *et al* (2007) suggest that the southern distribution limits of *L. hyperborea* and *Saccharina latissima* have not shifted in response to ocean warming over 50 years, based on historical surveys of intertidal habitats. It is very likely that kelp bed biomass and productivity will be diminished under warmer, stormier conditions (Krumhansl & Scheibling 2012a), although direct measurements of kelp bed structure, biodiversity, productivity, detritus production and export, and resistance and resilience to perturbation along a regional scale temperature gradient along the north-east Atlantic coastline are lacking. Comparative experimental work along regional scale temperature gradients is a promising approach in climate change ecology and can yield critical information on the mediation of ecological processes by ocean climate (Wernberg *et al* 2010, 2012). Comparative kelp research along a regional scale temperature gradient along Western Europe, spanning from Portugal (average sea temperature approximately 16°C) to Norway (average sea temperature approximately 8°C), would significantly enhance our understanding of climate change impacts on kelp bed structure and functioning.

Finally, two key knowledge gaps concerning the climate change ecology of kelp beds. Firstly, little is known of the capacity of local kelp populations to acclimatise or even adapt to climate mediated change. It is clear that kelp populations can maintain physiological processes under a wide range of environmental conditions through local adaptation (e.g. Delebecq *et al* 2013), but the rate at which kelp species can respond to rapidly changing temperatures and other localised stressors is unclear. Secondly, seaweed populations can be particularly susceptible to short term extreme warming events (Dayton & Tegner 1984; Smale & Wernberg 2013; Wernberg *et al* 2013), which may increase in magnitude and frequency as a consequence of anthropogenic climate change (Jentsch *et al* 2007; Feng *et al* 2013). Short-term climate variability may pose a greater threat to kelp populations at lower latitudes (i.e. towards range edges) than those within mid-latitude temperate regions. For example, southerly-distributed kelp beds off Spain and Portugal, which are subjected to environmental variability driven by the strength of coastal upwelling, comprise edge-of-range species with dynamic distributions (Fernandez 2011; Tuya *et al* 2012). Anomalous warming events also have the potential to cause step-wise changes in the structure and functioning of kelp beds in mid-latitude systems, and greater understanding of the resistance and resilience of kelp populations and their associated communities to such events is of ever-growing importance.

1.2.2 Top down – Grazers

Overgrazing by invertebrate herbivores, particularly sea urchins, can decimate kelp forests and cause phase shifts from structurally and biologically diverse habitats to depauperate ‘barrens’ (reviewed by Steneck *et al* 2002). Sea urchin populations are kept in check by a wide range of predators, including lobsters (Ling *et al* 2009), fish and sea otters (Estes & Duggins 1995), and by disease outbreaks (Scheibling 1986). Conversely, sea urchin population booms have been attributed to overfishing of sea urchin predators (Jackson *et al* 2001; Ling *et al* 2009), climate change (Ling 2008) and episodic recruitment events (Hereu *et al* 2004; Valentine & Edgar 2010). Following the formation of urchin barrens, a complex suite of interacting factors and feedback mechanisms affect the persistence of barrens and the likelihood of kelp bed recovery towards a pre-perturbed state (Norderhaug & Christie 2009). In the North Atlantic, the green sea urchin *Strongylocentrotus droebachiensis* has deforested extensive areas of kelp forest in eastern Canada (Mann 1977), Iceland (Hjorleifsson *et al* 1995) and northern Norway (Leinaas & Christie 1996), with major consequences for ecosystem structure and functioning (Steneck *et al* 2002). At lower latitudes, the importance of grazing by the purple sea urchin *Paracentrotus lividus* on macroalgal assemblages has been recognised along Mediterranean and Atlantic coastlines (Bulleri *et al* 1999; Hereu *et al* 2004; Tuya *et al* 2012).

In the UK and Ireland, the extent of deforestation by urchin grazing is generally restricted and patchy, although heavily grazed areas are more common in Scotland. Urchin grazing can certainly be important in setting local distributions of macroalgae, including kelps. Some of the earliest grazing work was conducted in the Isle of Man (Jones & Kain 1967), which showed that the edible sea urchin *Echinus esculentus* may determine the lower depth limit of *Laminaria hyperborea* stands through intense grazing of young sporophytes. Similarly, *Paracentrotus lividus*, which is relatively common along the west coast of Ireland, influences the distribution of macroalgae within Lough Ine through grazing activity (Norton 1978; Kitching & Thain 1983). The green sea urchin *Strongylocentrotus droebachiensis*, which is only found in the north of Scotland, may also cause restricted patchy deforestation, but extensive barren formation has not been attributed to this species.

Unlike many other temperate regions of the world, including Nova Scotia, the Gulf of Maine, eastern Australia, Alaska and Northern Japan (reviewed by Steneck *et al* 2002), there is little evidence for the formation of extensive, widespread sea urchin barrens off the UK and Ireland. Some of the most dramatic impacts of sea urchin grazing have been documented in regions where sea urchin predators, such as large lobsters (Ling *et al* 2009) and sea otters (Estes & Duggins 1995), have recently been removed through human activity. The consequent trophic cascade effects have, in some cases, led to widespread destruction of kelp forests and diminished biodiversity. In the UK and Ireland, apex predators (especially large finfish) have been overfished for centuries, so that large predatory crabs have become the dominant predators in many coastal ecosystems. As such, the likelihood of sea urchin population explosions resulting from removal of apex predators is probably low (see Sivertsen 2006, for Norwegian context). However, trophic interactions in kelp forests off the UK and Ireland are poorly understood and targeted research is required to address the level of threat posed by top down processes, which will be influenced by environmental change in complex and non-linear ways.

1.2.3 Harvesting and cultivation

The demand for kelp for human consumption, alginate production, aquaculture feed and (potentially) biofuel has increased in recent decades and will almost certainly continue to grow. Direct removal of kelps has major implications for kelp population structure, whole community dynamics and wider ecosystem functioning (Christie *et al* 1998; Vásquez 2008;

Krumhansl & Scheibling 2012a). There is some evidence to suggest that, due to the rapid recruitment and growth of kelps and their associated species, industrial-scale wild harvesting of kelps can be achieved sustainably. For example, in both Norway and Chile some 130,000 to 200,000 tonnes are extracted annually, and have been for some time (Vásquez 2008; Vea & Ask 2011). However, while a limited natural harvest may be sustainable if properly managed with appropriate fallow periods, the potential for impact on the other services provided by kelp may be considerable. Although kelps recruiting into harvested areas may reach pre-perturbed densities and sizes within a few years, their associated assemblages may take considerably longer to recover (Christie *et al* 1998). Kelp harvesting also negatively impacts the abundance of gadoid fishes and reduces the area of habitat preferred by foraging seabirds (Lorentsen *et al* 2010), for example.

Within the UK and Ireland, the potential for kelp biomass to be used for conversion to biofuels has reignited interest in large scale kelp production. A realistic contribution to energy markets through bioethanol production may require more kelp than can be harvested from natural habitats, prompting efforts to develop methods of farming kelp. Mariculture of kelps is commonplace in Asia, particularly in China, where demand for seaweeds for human consumption is high. It is clear that kelp farming can impact local patterns of water movement, and may cause organic enrichment of sediments and anoxia (Krumhansl & Scheibling 2012a). However, many researchers are championing integrated aquaculture practises that use kelps as bio-filters within multi-trophic farming operations (Neori *et al* 2004; Troell *et al* 2009). Within the UK context, the Crown Estate recently commissioned an independent investigation into the wider ecological effects of proposed seaweed mariculture off the west coast of Scotland (Aldridge *et al* 2012). Using ecosystem-based modelling approaches, the authors concluded that; *“the effects of the proposed farming activity on nutrient concentrations are expected to be 'marginally significant'......and “might become 'certainly significant'......The observable effects of nutrient removal would be a lower nutrient concentration in the water, decreased productivity and energy fluxes through the pelagic system, decreased flux of organic material to the seabed, and subtle alteration to community structure.”* (Aldridge *et al* 2012). Large scale kelp production, through both wild harvesting and mariculture, has the potential to impact kelp populations and associated benthic communities, and wider ecosystem structure and functioning. While it is recognised that a conservative ecosystem-based management approach is a pre-requisite for achieving sustainable production, the baseline knowledge on the structure and functioning of kelp ecosystems at regional scales needed to underpin such an approach is currently lacking.

1.3 Data on UK kelp

The purpose of this project is to define and validate indicators of the condition of kelp habitats and kelp populations on sublittoral rock that are responsive to relevant human pressures. In order to achieve this aim, the approach adopted here is to use existing data on kelp distributions and abundance around the UK to derive quantified statistical relationships with natural- and human-influenced environmental variables. This would establish the basis against which MSFD Descriptor 1 can be judged: “*1 - Biological diversity is maintained. The quality and occurrence of habitats and the distribution and abundance of species are in line with prevailing physiographic, geographic and climatic conditions.*” The data needed for this process had to meet adequacy criteria, particularly in relation to sampling effort and survey methods. Ad hoc species records, important and useful for establishing species ranges, often lack adequate information on methods and sampling methods, and are therefore difficult to incorporate in analyses. In this Section, the main sources of data on kelp abundance and distribution around the UK are reviewed.

1.3.1 Historical data – The Institute of Seaweed Research

The Institute of Seaweed Research (ISR) archives (now stored by Scottish Association for Marine Science) comprised a catalogued series of reports on surveys and other research, along with a number of annotated maps from surveys, such as that shown in Figure **1.6**, that contain much information that is only presently available in paper form. This resource deserves making more available in digital form, either as simple scanned PDFs or as transcribed data tables and GIS-based mapping files. The data would give an important insight into kelp habitat extent and biomass in the UK in the mid-20th century.

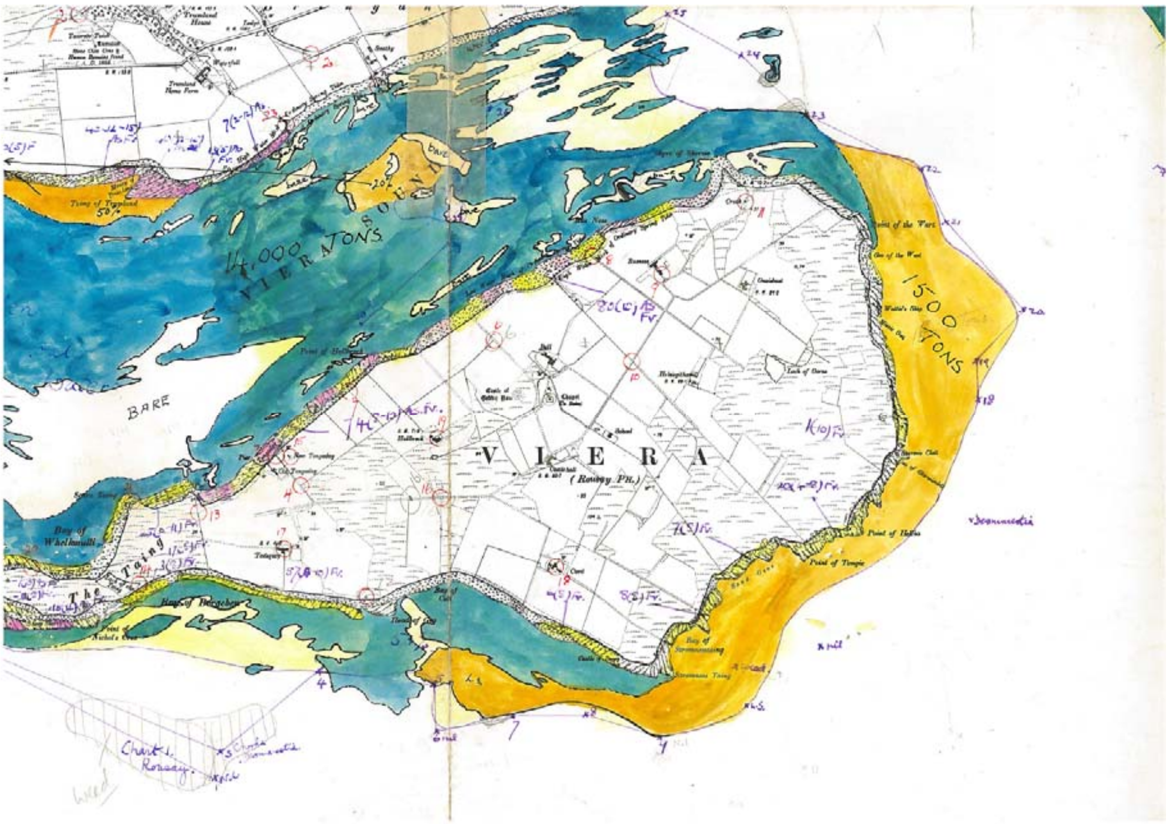

Figure 1.6. Kelp mapping by the Institute of Seaweed Research. Maps were created using aerial photography combined with surveys using spring grabs from a small vessel.

More usefully for comparison with modern estimates of kelp extent and biomass, scientists at the ISR used their survey data to estimate biomass totals for broad regions of Scotland, as part of the effort to identify exploitable resources for industrial extraction (Figure 1.7 and Figure 1.8).

Comparison of the tonnage estimates for intertidal seaweed resources with those estimated using GIS-based habitat modelling gives broadly similar values. The ISR estimated a total of 123,000t of *Ascophyllum nodosum* on the coasts of the Outer Hebrides, while a study by Burrows *et al* (2010) using surveys done in the summer of 2010 gave an estimate of 171,000t of *Ascophyllum* for the same area. The similarity of the biomass estimates from the two periods gives some confidence in the usefulness of the earlier estimates.

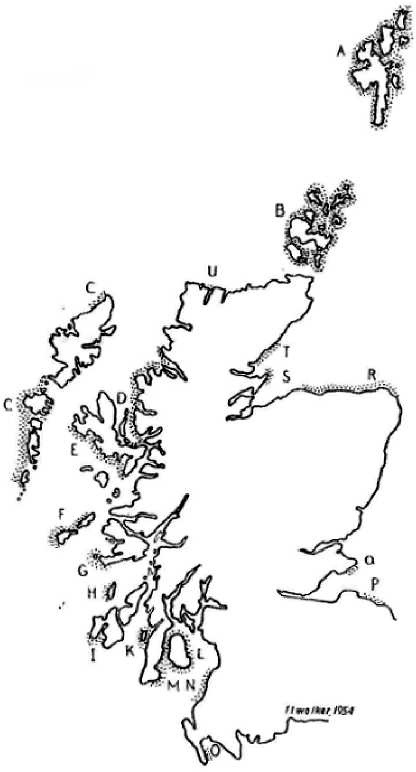

Figure 3. The distribution of the Laminariaceae around the coast of Scotland. The key to location, quantity, etc., is Table 2.

Figure 1.7. Distribution of kelp around Scotland from surveys by the Institute for Seaweed Research between 1946 and 1954, from Walker (1954)

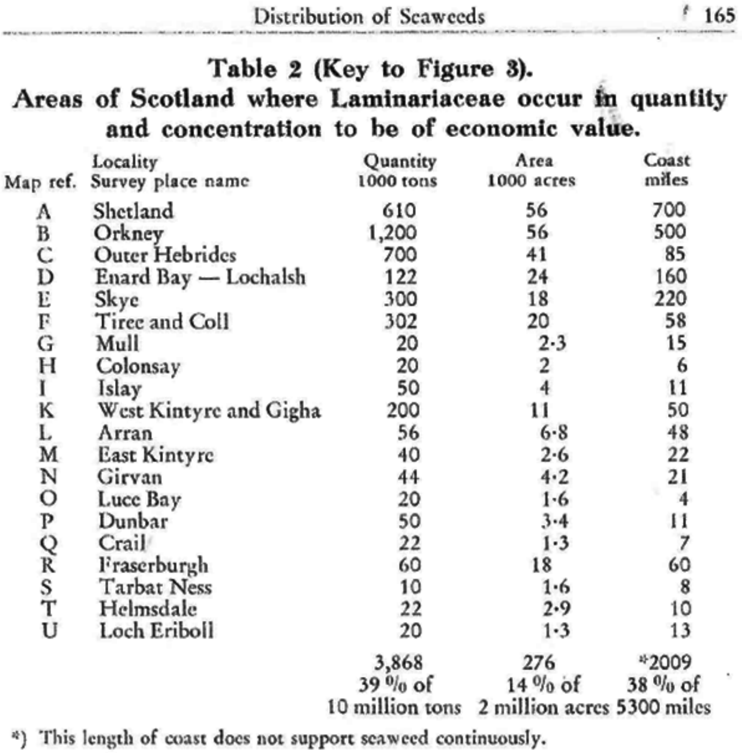

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Table 2 (Key to Figure 3).
Areas of Scotland where Laminariaceae occur in quantity and concentration to be of economic value.

Map ref.	Locality Survey place name	Quantity 1000 tons	Area 1000 acres	Coast miles
A	Shetland	610	56	700
B	Orkney	1,200	56	500
C	Outer Hebrides	700	41	85
D	Enard Bay — Lochalsh	122	24	160
E	Skye	300	18	220
F	Tiree and Coll	302	20	58
G	Mull	20	2·3	15
H	Colonsay	20	2	6
I	Islay	50	4	11
K	West Kintyre and Gigha	200	11	50
L	Arran	56	6·8	48
M	East Kintyre	40	2·6	22
N	Girvan	44	4·2	21
O	Luce Bay	20	1·6	4
P	Dunbar	50	3·4	11
Q	Crail	22	1·3	7
R	Fraserburgh	60	18	60
S	Tarbat Ness	10	1·6	8
T	Helmsdale	22	2·9	10
U	Loch Eriboll	20	1·3	13
		3,868 39 % of 10 million tons	276 14 % of 2 million acres	*2009 38 % of 5300 miles

*) This length of coast does not support seaweed continuously.

Figure 1.8. Estimated tonnage of kelp around Scotland by the Institute for Seaweed Research in the late 1940s and early 1950s, from Walker (1954).

1.3.2 The Marine Nature Conservation Review

Kelp habitats were extensively surveyed by the Nature Conservancy Council (NCC) and the JNCC's Marine Nature Conservation Review (MNCR), predominantly in the 1980s and 1990s. Surveys were done by divers following well developed methods and protocols (Connor & Hiscock 1996; Hiscock 1996), and generally to a high level of taxonomic resolution. The resulting data is now held in the JNCC's Marine Recorder database, and is publicly available via the JNCC website. Surprisingly, this data resource has not been widely used by academic researchers, but the spatial trends in the kelp species and associated biota have been the subject of a study by Burrows (2012). This study quantified the already observed patterns with major environmental drivers, such as the progressive loss of kelp from deep water when moving into areas of high sediment load (Hiscock 1985). This study plans to use an extension of this approach to establish the baseline for kelp under the MSFD Descriptor 1: '*Biological diversity is maintained. The quality and occurrence of habitats and the distribution and abundance of species are in line with prevailing physiographic, geographic and climatic conditions.*' Predictions of kelp extent and biomass, and the diversity of kelp-associated biota can be made using the statistical relationships with major environmental predictors using this dataset.

1.3.3 British Phycological Society mapping scheme

The British Phycological Society has maintained a mapping scheme recording presence data around the whole of the UK and Ireland since 1971 (Hardy *et al* 2003). This is a comprehensive and strongly validated dataset, maintained by experts. The data include most if not all of the MNCR records. The emphasis is on recording the presence of species at particular locations, and as such the data are very useful for analyses of species richness (Blight *et al* 2009). Aggregation of data into 10km grid squares does limit the utility of the information in regard to resolving small-scale habitat variability, such as that due to changes in wave exposure.

1.3.4 Other datasets

The National Biodiversity Network (NBN) Gateway (https://data.nbn.org.uk/) holds a wide range of species information. NBN holds 10,300 records for *Laminaria hyperborea* across 23 recording schemes from 16 environmental bodies (Table 1.1). There may be potential to expand the range of sources that provide data that can be effectively analysed together, but further work is needed to ensure comparability for collection and recording methods, including spatial and taxonomic resolution, measures of abundance and sampling effort.

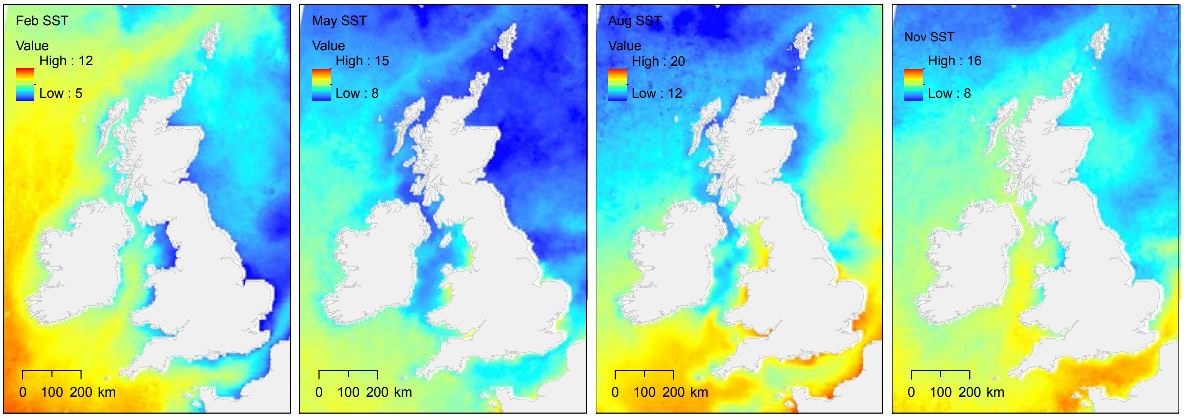
Table 1.1. Data sources for records of *L. hyperborea* on the NBN Gateway accessed 25/03/14

Data holders	Recording Scheme	Date added
Merseyside BioBank	North Merseyside Other Taxa (unverified)	25-Jan-2014
Outer Hebrides Biological Recording Project	OHBRP Fungi, Lichens and Lower Plants Dataset - Outer Hebrides	24-Sep-2013
Environmental Records Information Centre North East	ERIC North East non-sensitive species records	29-Jul-2013
Marine Conservation Society	Seasearch Marine Surveys	25-Jul-2013
Scottish Natural Heritage	Marine species data for Scottish waters held and managed by Scottish Natural Heritage, derived from benthic surveys 1993 to 2012	27-Jun-2013
Natural Resources Wales	Marine records from Skomer Marine Nature Reserve (MNR) Marine Monitoring Programme	02-May-2013
Natural Resources Wales	Marine records from Pembrokeshire Marine Species Atlas	02-May-2013
Natural Resources Wales	Marine data from Natural Resources Wales (NRW) Technical Support (Research & Monitoring) Contracts, Wales	02-May-2013
South East Wales Biodiversity Records Centre	NRW Regional Data : South East Wales Non-sensitive Species Records	05-Mar-2013
Yorkshire Naturalists' Union Marine and Coastal Section	Yorkshire Naturalists Union Marine and Coastal Section Records	20-Nov-2012
Marine Biological Association	DASSH Data Archive Centre expert sightings records	29-Oct-2012
Centre for Environmental Data and Recording	Marine Data from Northern Ireland	17-Jan-2012
Natural Resources Wales	Marine Intertidal Phase 1 species dataset from the Countryside Council for Wales 1996-2005	19-Dec-2011
Cumbria Biodiversity Data Centre	Norman and Florence Hammond records. Seawatch and coastal survey records.	26-May-2011
Porcupine Marine Natural History Society	Marine flora and fauna records from the North-east Atlantic	22-Oct-2010
Cofnod (North Wales Environmental Information Service)	NRW Regional Data: North Wales	13-May-2010
Marine Biological Association	DASSH Data Archive Centre volunteer survey data	26-Nov-2008
Marine Biological Association	DASSH Data Archive Centre volunteer sightings records	26-Nov-2008
Marine Biological Association	DASSH Data Archive Centre Statutory Agency and commercial marine surveys	26-Nov-2008
Marine Biological Association	DASSH Data Archive Centre academic surveys	26-Nov-2008
British Phycological Society	Seaweed data for Great Britain and Ireland	15-May-2008
Natural England	Marine Nature Conservation Review (MNCR) and associated benthic marine data held and managed by English Nature	22-Apr-2005
Joint Nature Conservation Committee	Marine Nature Conservation Review (MNCR) and associated benthic marine data held and managed by JNCC	22-Apr-2005

1.4 Environmental drivers and pressures

For the development of a kelp indicator of good environmental status, this study uses the relationships between species abundance and underlying patterns in predictor variables (see Sections 1.3 and 2.1). In this Section, the major gradients in environmental drivers for sublittoral communities in the UK are reviewed. Ocean colour and infrared radiation can be detected using satellite remote sensing of the ocean surface, and give synoptic views of the conditions of pelagic productivity and temperature around the UK. Maps of other drivers, such as wave exposure and tidal flow, can be derived from model outputs. The wider European and North Atlantic context of these conditions is also important, particularly when anticipating future effects of a changing climate and the potential for species range shifts into UK waters. Thermal niches for marine species have been moving poleward (eastwards in the English Channel) at widely varying rates around the UK since 1960, with the slowest rates of about 20km/decade in north-western regions, and much faster, up to 200km/decade, in the North Sea (Burrows *et al* 2011).

1.4.1 Temperature



(A) Feb (B) May (C) Aug (D) Nov

Figure 1.9. Sea surface temperature around the UK for (A) February, (B) May, (C) August, and (D) November; from averages over the period 2000-2006 obtained from the NASA Giovanni Data Portal (DAAC 2008).

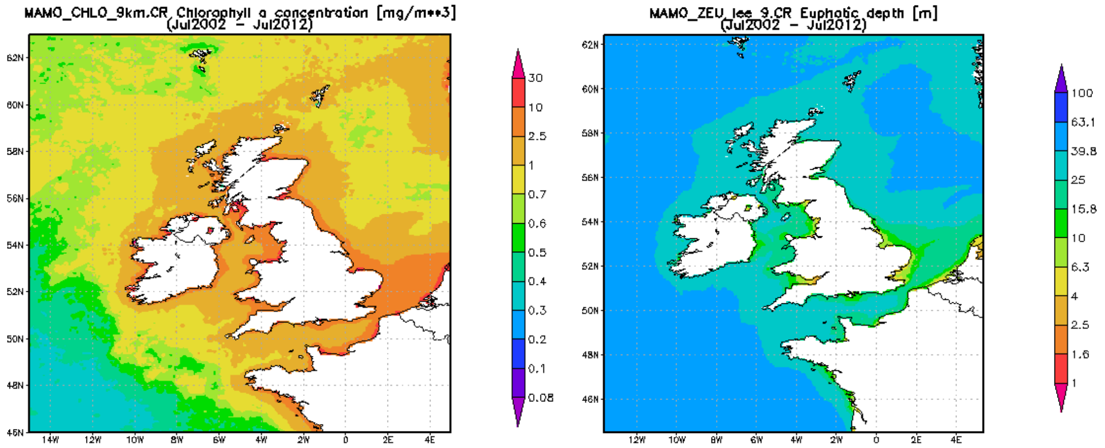
Temperature is a major driver setting geographic patterns of the distributions of marine species (Hutchins 1947; Vermeij 1978; Lüning 1990), although factors such as habitat type and extent and local hydrography can determine local abundance and be involved in setting range edges. Sea Surface Temperature (SST) derived from satellite images, is a widely available dataset which can be used for biogeographic studies.

Maps of average patterns of SST: winter (February), spring (May), summer (August) and autumn (November) show the general pattern of increasing temperatures from the south to the north around the UK (Figure 1.9). The axis of change varies from summer to winter, with February SST generally higher in the west (Atlantic and Irish Sea) than in the east (North Sea), and August SST warmer in the southern areas of both east and west coasts of the UK. Regional scale patterns also vary between summer and winter. Summertime patterns show evidence of stratification, mostly delineated by the presence of tidal fronts (Pingree & Griffiths 1978), such as the Islay and Celtic Sea fronts, separating stratified areas with warmer surface water from tidally well-mixed areas with cooler surface water. Locally cooler areas such as the north-west tip of Cornwall may also influence the relative proportions of cold and warm water species, and are important to note.

Most rocky shore species whose distributions reach their northern limits in the UK follow a distribution pattern similar to that of February SST, being most likely to be present in the west and south, and least likely in the north and east (Lewis 1964).

1.4.2 Water quality - Primary production and sediment load

Water quality results from the biomass of phytoplankton and the quantity of suspended material per unit volume. Both these properties influence the spectral characteristics of reflected sunlight, but are not always easy to distinguish. The presence of significant amounts of suspended sediments makes estimation of the chlorophyll concentration problematic. As a consequence, inshore coastal waters are considered differently from offshore oceanic waters (see, for example, Joint & Groom 2000). Notwithstanding such issues, the influences of phytoplankton biomass and suspended sediment on rocky shore biota are likely to be similar and for the purposes of developing indicators can be considered together. Increased phytoplankton and sediment increase the attenuation of light through the water, reducing the depth to which sufficient light penetrates to allow photosynthesis (‘compensation depth’ – where respiration of plants exceeds the photosynthesis). The euphotic depth is much reduced in areas of high sediment and phytoplankton, such as in the Bristol Channel, southern North Sea coasts and Liverpool Bay (Figure 1.10). The abundance of sublittoral macroalgae is much reduced in these areas (Burrows 2012).



(A) (B)

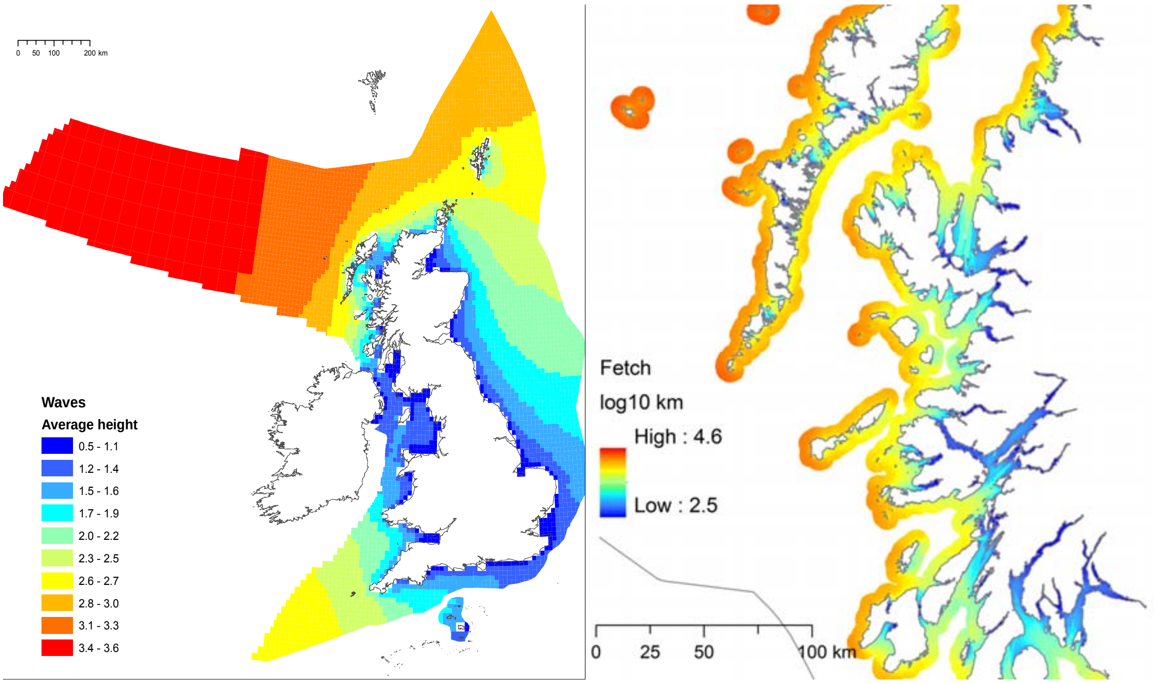
Figure 1.10. (A) Chlorophyll *a* concentration (mg/m^3), (B) Euphotic depth (m) in UK and Irish waters.

In this report, chlorophyll *a* concentrations are referred to as mg/m^3 in the full knowledge that these values are influenced by suspended sediment. The probable effects of suspended particulate material and phytoplankton on rocky sublittoral communities are due to a complex set of inter-related processes, probably due to a combination of shading of macroalgae, greater retention of larvae of suspension feeders and enhanced feeding conditions for this same group (Burrows 2012). Data shown ((A) **B**)) were obtained from the NASA MODIS Aqua satellite sensor, averaged over the period from July 2002 to July 2012. Coarser resolution (9km) web-available NASA satellite data may be better for comparison with benthic communities, since finer scale (1-4.5km) data may misrepresent the highly dynamic nature of phytoplankton concentrations at this scale. Current speeds may be such that patchiness at these finer scales does not persist for more than a few hours or days, while the benthic biota integrates responses to water column conditions over months and years. The data we propose to use is aimed at resolving differences on 10 to 100km scales in a primarily biogeographic or coastal cell context, and it is appropriate for MSFD purposes that environmental data with similar resolution is used. New techniques for local scale remote sensing, such as unmanned aerial vehicles flying under clouds, may shed light on the importance of small scale variation in pelagic productivity, but this knowledge is not yet available.

1.4.3 Wave height and wave exposure

Wave height is generally highest on open Atlantic-facing coasts, especially in the north and west of Scotland, west of Ireland and along the Western Isles and least along the coasts of the semi-enclosed Irish and North Seas and the English Channel (Figure 1.11A). Importantly, however, the influence of offshore oceanic waves is strongly modified by coastal topography. The local shelter offered by headlands and islands much reduces wave action, and very enclosed areas such as firths, channels and sea lochs have very little wave action (Figure 1.11B). Wave fetch, the distance over which winds blow before reaching any piece of land, is a good predictor of wave height. Wind waves follow well-understood physical laws, and wave heights and spectral characteristics (mixture of short-period and long-period waves) directly depend on the length of time that winds of specified velocities blow over the water surface. Even without including the more complex physics, simple indices of wave exposure have proved effective in predicting patterns in coastal ecosystems. Wave exposure indices range from relatively simple, such as a count of the number of sectors open to the sea (Baardseth 1970) to complex, using a sum of wave fetch values in all directions open to the sea, and weighted by the incidence and average speed of winds from those directions from local meteorological data (Thomas 1986). Fetch-only indices can perform as well as those including wind information, especially over areas where the wind pattern is relatively consistent (Burrows *et al* 2008).

In this study, a modelled measure of wave fetch (Burrows 2012) is used to establish the prevailing physical conditions at any MNCR survey site, and derive expectations for the composition of the community at that site. The index is based on the minimum distance (km) to the nearest land in each of sixteen 22.5°angular sectors, up to a maximum of 200km per sector. Given the wide range of values around the UK, from 1 to 32,000km, it has been most convenient to express these as log base 10 values from 0 to 4.5 (Figure 1.11B). The model has a spatial resolution of 200m, so may be well matched to the reported spatial resolution of MNCR surveys.



(A) (B)

Figure 1.11. (A) Offshore wave height around the UK showing wave height data from the UK Atlas of Marine Renewables (ABP Marine Environmental Research Ltd 2008), (B) Inshore wave exposure derived from wave fetch (<5km from the coast) based on inshore wave fetch estimated using a model described in Burrows *et al* (2012).

1.4.4 Other factors: Tidal flow, tidal range, geology

Tidal flow is relatively high in regions of restricted flow between larger water masses, such as the connections between the North Atlantic and North Sea through the Pentland Firth and between Orkney and Shetland. Headlands also present barriers and cause flows to accelerate around them. Flows are also high at the mouth of macro-tidal estuaries like the Bristol Channel and, to a lesser extent, the Solway Firth. A consequence of the latter effect is that there is a reasonably strong correlation between flow and wave exposure.

Effects of tidal flows on sublittoral species and communities are not well understood, although, where these are strong, greater flow may promote the growth of suspension feeders (Sanford *et al* 1994; Leonard *et al* 1998; Sanford & Menge 2001) with consequential effects on community structure. Potentially, the more important effects of tidal flows on rocky shore communities may be indirect. Sediment transport is driven by tidal flows, with rapid flows generally associated with coarser sediment or no sediment at all. The high flows around estuaries and in general in the southern North Sea have a strong influence on the amount of suspended material in the water (Figure 1.12A, and Figure 1.10B above), and this may have a stronger influence on attached animals and plants than the flow itself. Intertidal species are generally exposed to much greater flows and forces during breaking waves (Denny 1988) than those exerted by the relatively gentle tidal flows.

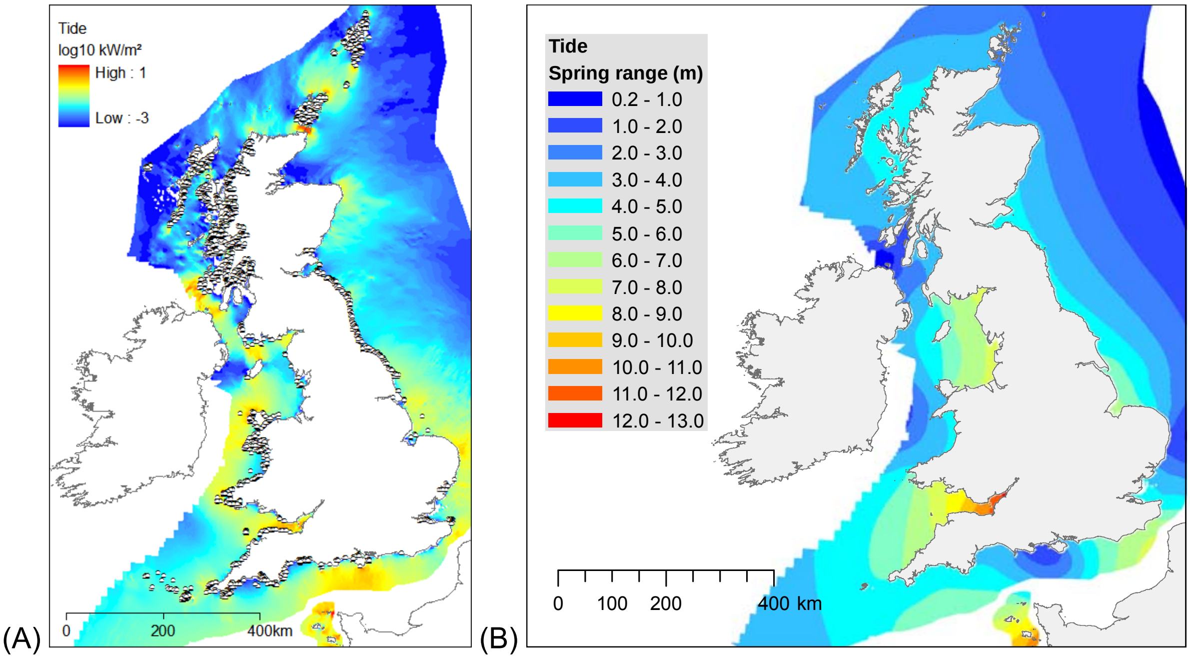


Figure 1.12. (A) Estimated tidal power at the seabed (kW/m^2) from a 1.8-km resolution model produced by the Proudman Oceanographic Laboratory, and (B) spring tidal range (m), taken from the UK Atlas of Marine Renewables (ABP Marine Environmental Research Ltd 2008).

Areas (microtidal) of locally restricted tidal ranges (with a spring tidal range of less than 1.0m such as in the Sound of Jura and in the area from Swanage to the Isle of Wight, Figure 1.12) contrast with macrotidal areas, such as the Bristol Channel and eastern Irish Sea. Reduced tidal ranges may result in less available habitat for intertidal species, and potentially reduced diversity.

Rock type is not considered by rocky shore ecologists as a major driver of species distributions, except for very soft rock such as chalk and shale (such as in Dorset). Soft rock does have a distinctly different biota, dominated by red algae and generally much reduced from that seen on hard rock in similar conditions due to higher erosion rates dislodging sessile species (Connor *et al* 2004), and may be involved in setting range limits in the English Channel (Herbert & Hawkins 2006; Keith *et al* 2011). Differences in community composition among different rock types from limestone ledges, conglomerates, granite, slate, for example, are less well documented but most likely reflect the differences in surface complexity (Frost *et al* 2005). The reduction in available habitat for rocky shore species on sediment-dominated coasts, such as along the eastern Irish Sea coasts of North Wales, Lancashire and Cumbria, and from Kent to Flamborough Head, may further reduce the diversity of rocky shore species.

1.5 Statistical methods

The relationships between kelp abundance on categorical abundance scales and environmental variables, and the relationships between species richness of kelp associated species and kelp abundance and drivers, allow for quantitative predictions for kelp abundance and kelp habitat species diversity in different combinations of environmental conditions around the UK. These expectations represent the foundation for judging environmental status under the MSFD Descriptor 1 '*Biological diversity is maintained. The quality and occurrence of habitats and the distribution and abundance of species are in line with prevailing physiographic, geographic and climatic conditions.*' Statistical models of changes in diversity and abundance with measured (or estimated) values of environmental variables are an objective way to precisely define the '*in line with*' concept embedded in Descriptor 1. Such statistical models are built from patterns in existing data, and are unbiased by 'expert opinion': an advantage inasmuch as they are not coloured by, for example, undue focus on particular taxa imposed by particular marine experts. That said, careful construction of models and interpretation of the results is needed to ensure that the responses to environmental and human pressures are identifiable.

Ordinal logistic regression is used to relate categorical SACFOR kelp abundance data from the MNCR and simple multiple regression is used to relate diversity measures to the values for variables representing environmental drivers outlined in Environmental drivers and pressures. In principle, it is anticipated that the ecological status of kelp forests would be best judged by the observed abundance and diversity of species relative to the statistically derived expectations at local and regional scales. Low abundance of kelp and/or low diversity of kelp-associated species relative to expectations would point towards poor ecological status.

2 Developing indicators of Good Environmental Status for the Marine Strategy Framework Directive

This section aims to describe the process by which indicators of Good Environmental Status (GES) for sublittoral rock communities based on the condition and composition of kelp and kelp parks were developed. The process begins with the identification of the relevant criteria against which environmental status should be judged, then assesses which environmental parameters may predict kelp distributions, and finally develops the statistical models against which GES may be judged.

2.1 MSFD Descriptors

Good Environmental Status for the MSFD relies on eleven descriptors of which two are relevant for the relatively undisturbed and unexploited habitats on shallow sublittoral rock. The most relevant descriptor is:

1. Biological diversity is maintained. The quality and occurrence of habitats and the distribution and abundance of species are in line with prevailing physiographic, geographic and climatic conditions.

Other descriptors are linked, and a particular example is that dealing with eutrophication, known to impact on attached macrolagae

5. Human-induced eutrophication is minimised, especially adverse effects thereof, such as losses in biodiversity, ecosystem degradation, harmful algal blooms and oxygen deficiency in bottom waters.

The approach to the development of the kelp and kelp park indicators has been taken from the principle embodied in Descriptor 1 above: '*The quality and occurrence of habitats and the distribution and abundance of species are in line with prevailing physiographic, geographic and climatic conditions*', while also seeking to address the Commission Decision (2010/477/EU[1]) indicator 1.6.1 – Condition of the typical species and communities - for sublittoral rocky habitats in UK waters. Thus, GES, within this context, would reflect that species are present as expected, given the presence of the kind of habitats where they normally occur, and the prevailing physical conditions. The availability of extensive data on the abundance of sublittoral rock species allows an objective analysis of patterns in respect to spatial gradients and temporal changes that gives a baseline against which observed status can be compared, and thereby provides the supporting evidence for indicators of kelp habitats comprising data in the form of kelp abundance as assessed using Marine Nature Conservation Review (MNCR) methods (see Section 2.2.1).

2.2 The distribution of kelp and associated species along environmental gradients in the UK

Patterns of abundance of kelp and associated species around the UK have recently been quantified in a study by Burrows (2012), which focussed on changes in abundance along gradients of wave exposure and depth, and how the abundance is modified by local conditions of tidal flow and the concentration of phytoplankton and suspended sediments in the overlying water column. Burrows (2012) produced a numerical index of wave exposure for the entire coast of the UK and Ireland at a 200m scale up to 5km offshore, and enabled a

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[1] http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2010:232:0014:0024:EN:PDF

quantitative analysis of patterns along this and other gradients. The production of this wave exposure index, based on estimated wave fetch (the distance over which wind blows before reaching a particular location), and geographical patterns of wave exposure are described more fully in Section 2.3.2.

The patterns of species distributions along environmental gradients form the basis of our understanding with regard to the stipulation of "*in line with prevailing (…) conditions*" within MSFD Descriptor 1. Most of the patterns described by Burrows (2012) have long been recognised, such as the restriction of kelp communities to shallow depths in water of greater light attenuation (Kain 1977) resulting in the shoaling of the lower limit of kelp and foliose algae up the Bristol Channel towards the mouth of the Severn along a gradient of increasing turbidity (Hiscock 1985). Applying statistical models to a large available dataset, diver surveys done as part of the MNCR, has made possible the visualisation and quantification of these patterns and trends, which are presented within this report (Figure 2.1, Figure 2.2, Figure 2.3, Figure 2.4). The figures shown for each species in this section were produced as surface plots based on predicted probabilities of percentage cover of the substratum. Probabilities plotted are those associated with abundance reaching (and exceeding) the 'Common' abundance category (see Appendix 1, >10% cover for kelp). In this section (2.2) probabilities were predicted using ordinal logistic regression of ranked categories of abundance, using region, depth and wave fetch classes (split into 5m and 0.5 $\log_{10}$ fetch units respectively) as predictor variables and using the analysis described in Burrows (2012). This analysis is subsequently extended to include the environmental variables of sea surface temperature (SST) and chlorophyll *a* concentrations to give meaningful predictive models that can be applied at a UK scale (see Section 2.4). Justification for the choice of these environmental parameters is provided in Section 1 of this Report, but, in essence, these are the best known available predictors of kelp distributions around the UK.

2.2.1 The Marine Nature Conservation Review dataset

Sublittoral rock habitats were surveyed using SCUBA diving (Connor & Hiscock 1996). Divers swam along a transect assessing the abundance of species over an extended area of habitat within a recorded depth range, most likely along a line spanning <200m of habitat given the time limitation of a single SCUBA dive. Species abundance was estimated as population density or percentage cover, then recorded as a categorical abundance value on the MNCR SACFOR scale (Superabundant, Abundant, Common, Frequent and Rare, see Appendix 1) (Connor *et al* 2004). Surveys were generally completed in a day over several dives over different depth ranges at each location. Absences of species were not recorded: no distinction could be made between true absence (not seen – a true negative) and a lack of a positive record (false negative). All negative records were treated as absences in further analyses.

2.2.2 Wave exposure and depth as the major axes of variation in shallow sublittoral communities

The major kelp species around the UK described in Section 1.1.2 have distinct characteristic distributions along wave exposure and depth gradients (Figure 2.1 to Figure 2.3).

Sugar kelp *Saccharina latissima* (also known as sea belt or *smeartan* or *mìlearach* in Scottish Gaelic), is most likely to be found in wave sheltered areas (Figure 2.1A; 1.5-3.0: green, yellow, orange and red areas on Figure 2.6), and most abundant in the top 5m. The annual kelp species *Saccorhiza polyschides* (sometimes referred to as furbellow) is found in intermediate wave exposure (Figure 2.1B; 2.5-3.5: yellow to light blue areas on Figure 2.6), and most abundant in the top 10m.

The dominant kelp in wave exposed environments is the multiannual *Laminaria hyperborea* (Figure 2.2A; 3.0->4.0: light blue to mauve areas on Figure 2.6) and is generally found at greater depths than *S. latissima* or *S. polyschides*. The other notable kelp species on wave exposed sublittoral rock is *Alaria esculenta* (bladderlocks, dabberlocks or *mircean* in Scottish Gaelic), most likely found in the most extreme wave exposed environments (Figure 2.2B; >3.5, dark blue and mauve areas on Figure 2.6). *A. esculenta* is also commonly recorded during intertidal surveys, being recorded as present in 30% of intertidal surveys around Scotland made between 2002 and 2009 by SAMS survey teams.

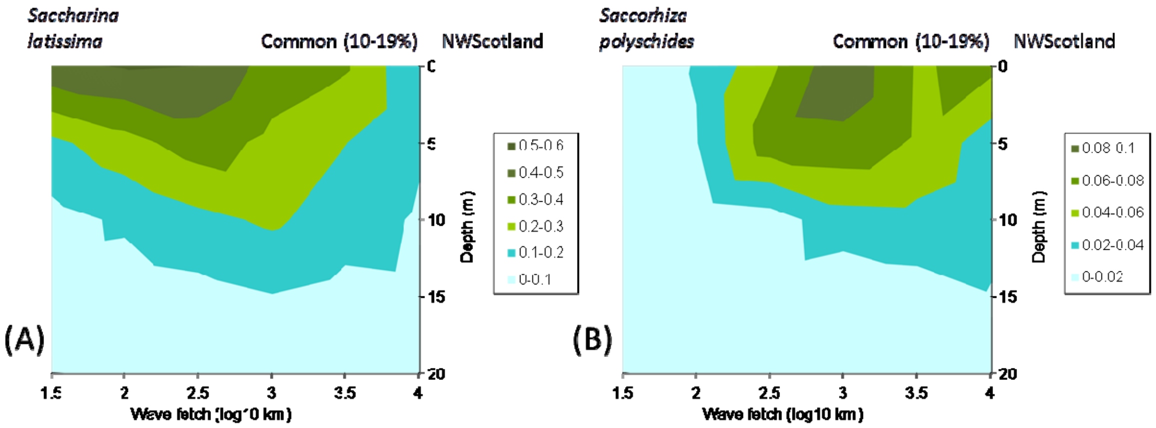


Figure 2.1. Distributions of the kelps (A) *Saccharina latissima* and (B) *Saccorhiza polyschides* along gradients of wave fetch and depth in north-west Scotland. Shading indicates the probability of finding each species with at least 10% cover of the seabed ('Common' on the SACFOR scale), derived from ordinal logistic regression models based on UK MNCR data, and scaled 0 - 0.6 for *S. latissima* and 0 - 1 for *S. polyschides*. From Burrows (2012).

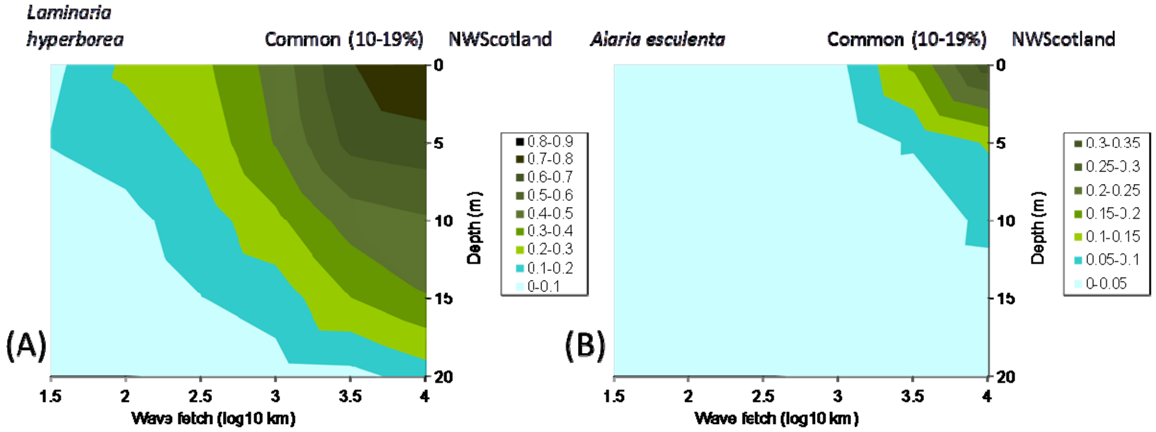


Figure 2.2. Distributions of the kelps (A) *Laminaria hyperborea* and (B) *Alaria esculenta* along gradients of wave fetch and depth, showing probability of finding each species with at least 10% cover ('Common' on the SACFOR scale) in north-west Scotland, from ordinal logistic regression models of MNCR data, scaled 0 to 0.9 for *L. hyperborea* and 0 - 0.35 for *A. esculenta*. From Burrows (2012).

Laminaria digitata (tangle or *liadhag, stamh* in Scottish Gaelic) is a shallow water species found across all levels of wave exposure, but mostly restricted to the top 5m or less (Figure 2.3), and the most likely kelp to be recorded in low tide surveys, present in 60% of SAMS intertidal surveys around Scotland.

The only other UK native kelp species, the warm-water kelp *Laminaria ochroleuca* is limited to southern England and was not present in enough MNCR surveys to warrant analysis of trends in distribution. The non-native kelp *Undaria pinnatifida* is increasing in abundance but also lacking data to allow visualization of trends.

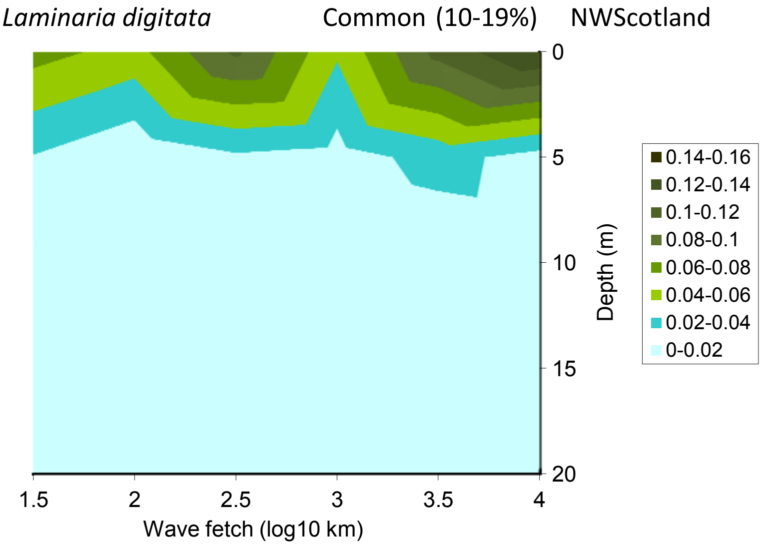


Figure 2.3. Distribution of the kelp *Laminaria digitata* along gradients of wave fetch and depth. Shading indicates the probability of finding each species with at least 10% cover of the seabed (‘Common’ on the SACFOR scale), derived from ordinal logistic regression models. Shading applies to habitats in north-west Scotland and scales from 0 - 0.16. From Burrows (2012).

2.2.3 Species associated with kelp distributed along wave action and depth gradients

Other important species of sublittoral rock have characteristic distributions along wave exposure and depth gradients, shown by plots of the likelihood of finding the species as being at least ‘Common’ on the SACFOR scale (Burrows 2012). Many species of red algae have a similar distribution across depth and wave exposure to that of *Laminaria hyperborea* (Figure 2.4B), with some species extending deeper than the kelp, such as *Bonnemaisonia asparagoides, Phycodrys rubens* and *Dictyota dichotoma*. Other red algae are found mostly in shallower depths, including *Chondrus crispus* and *Palmaria palmata*. *Desmarestia aculeata* is more likely to be abundant in less wave-exposed environments (Figure 2.4B; 2.0-3.0, light brown to green areas of Figure 2.6).

The geniculate or articulated coralline algae are more abundant in extremely wave exposed habitats (Figure 2.4C; >3.5, dark blue to mauve areas in Figure 2.6). Encrusting or non-geniculate corallines are apparently much more sensitive to wave action, each being found in a relatively confined range of fetch values (Figure 2.4D).

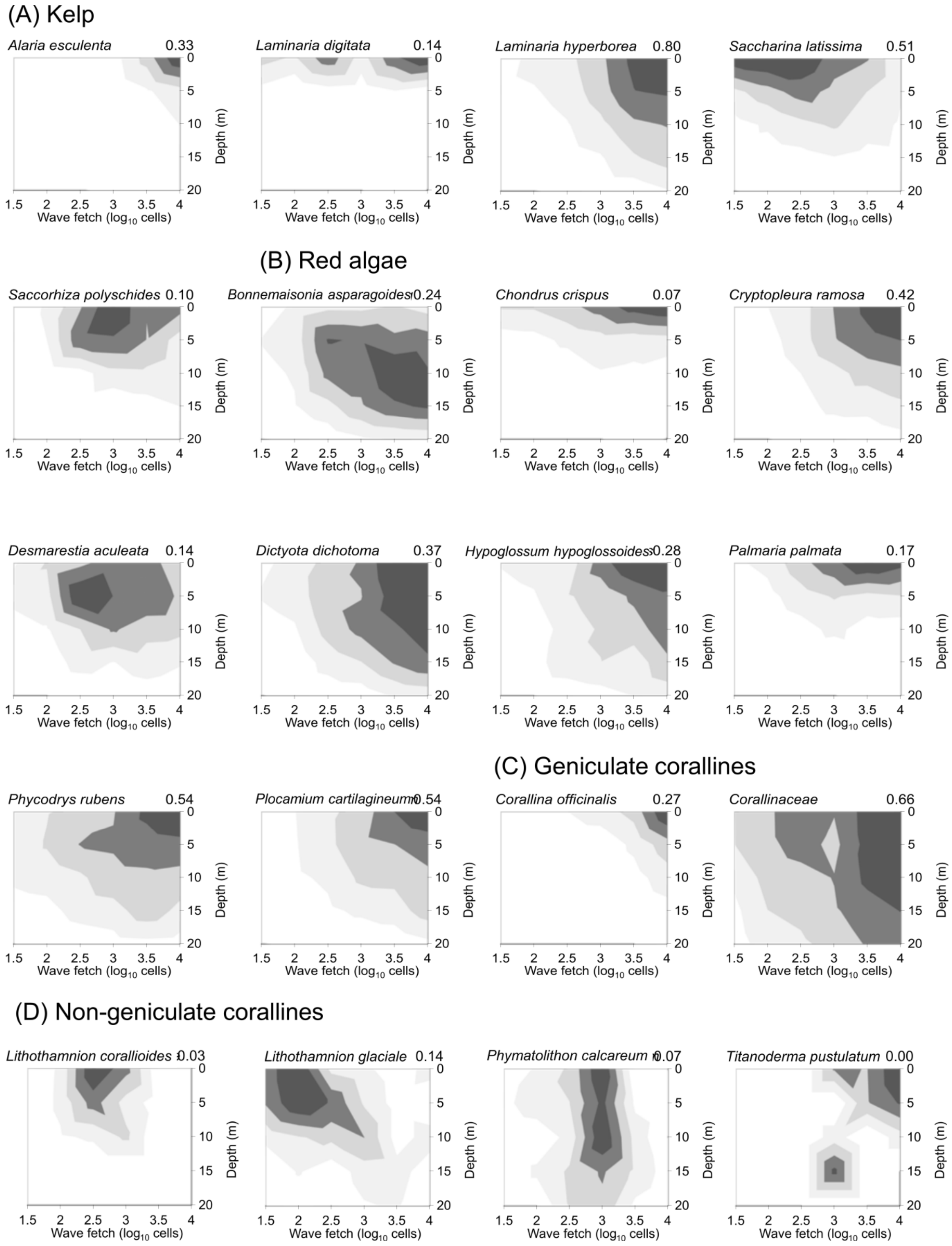
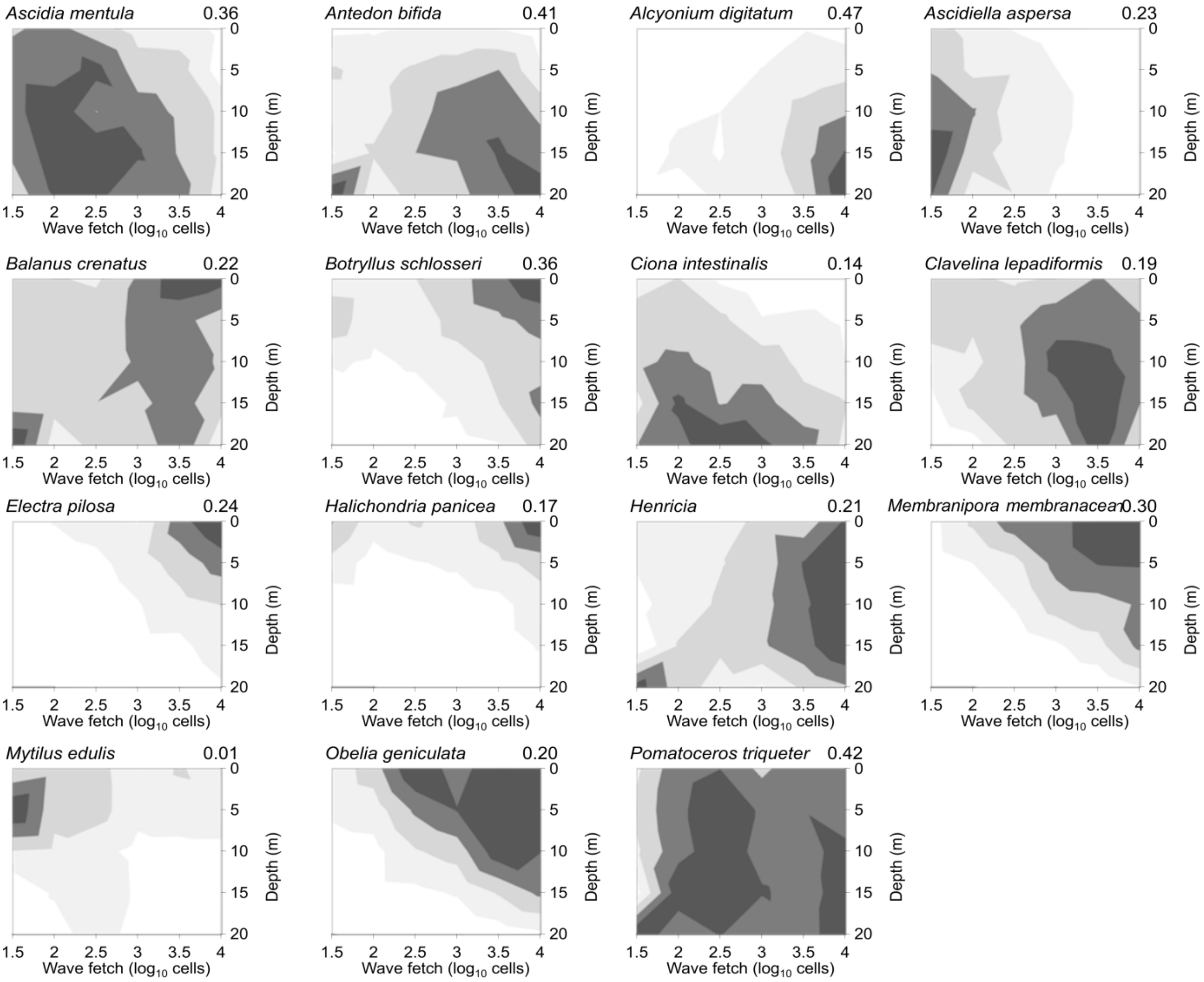
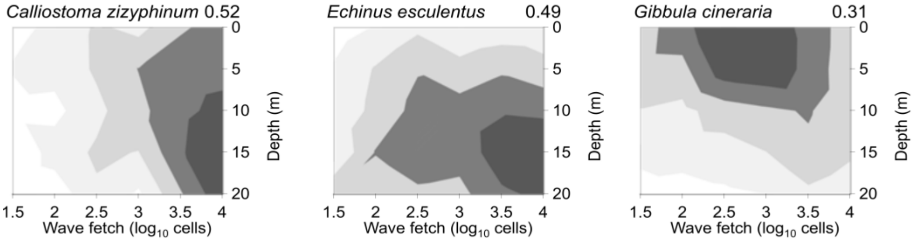


Figure 2.4. Distributions of (A) kelps, (B) red algae, (C) geniculate coralline algae, and (D) non-geniculate corallines along gradients of wave fetch and depth in the UK. Kelp species are displayed for comparison alongside the kelp-associated species to highlight potential environmentally driven species associations.

(E) Suspension feeders



(F) Grazers



(G) Predators

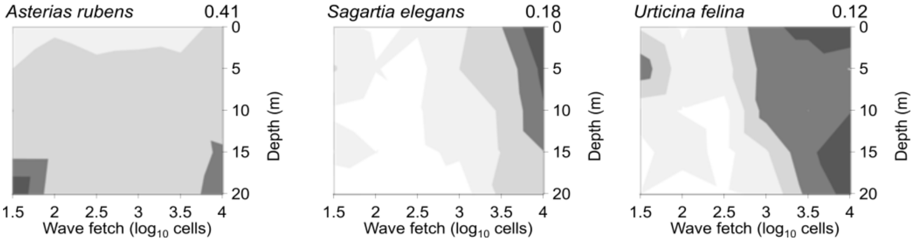


Figure 2.4 (continued) Distributions of (E) suspension feeders, (F) grazers, and (G) predators along gradients of wave fetch and depth in the UK.

The fauna of sublittoral rocky habitats generally extends much deeper than the macroalgae (Figure 2.4E to G), but shows a corresponding shift in community composition along wave exposure and depth gradients. More species are associated with moderate to high wave exposure than low wave exposure, with only the tunicates *Ascidia mentula* and *Ciona intestinalis* being confined to greater depths in low wave energy environments. *Obelia geniculata*, *Botryllus schlosseri* , *Mebranipora membranacea* and *Henricia* all show the opposite pattern, being more likely to be found in shallower depths in high wave energy environments (Figure 2.4E). Some species, like the anemones *Urticina felina* and *Sagartia elegans*, with a passive mode of feeding may rely on water motion to bring their food supply in their preferred wave-exposed habitats (Figure 2.4G), and may benefit from mortality to other species due to wave-induced disturbance damage.

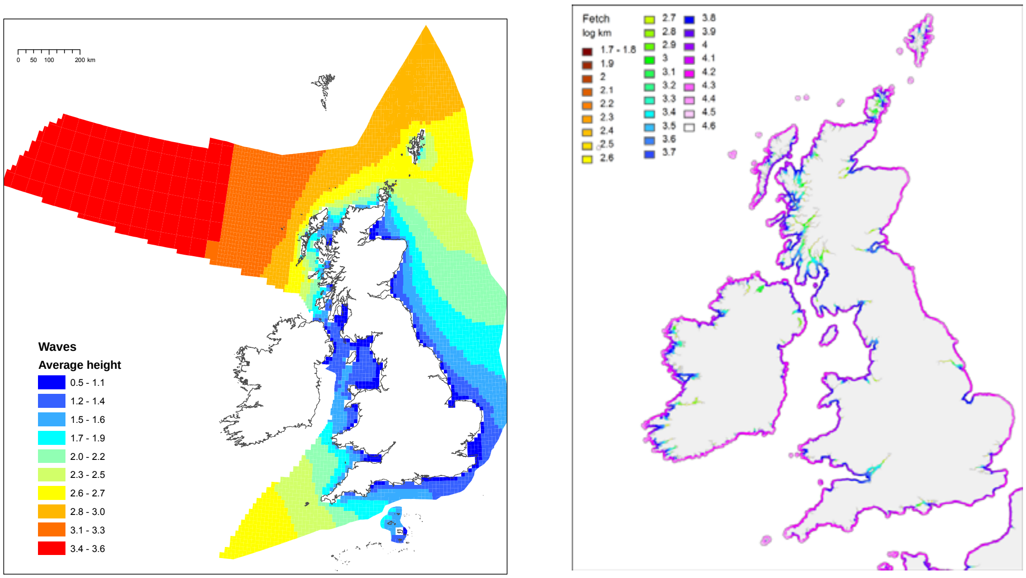
2.3 Characterizing habitats for kelp species

The recommendations presented for assessment of ‘Good Environmental Status’ for kelp habitats are based on establishing what characteristics those habitats should normally have for site-specific combinations of environmental conditions. This embodies the principle behind MSFD Descriptor 1: “*prevailing […] conditions*” and allows the identification of a baseline against which deviations in ecological status, associated with particular human pressures, can be measured. This study aims to characterise the four parameters to underpin the proposed methodological approach for the kelp indicators: (i) the likely presence of each of the five major UK kelp species at a site; (ii) the likely abundance of those species as a function of depth at a site; (iii) the maximum kelp depth, below which kelp abundance is less than 20% cover; and (iv) the maximum kelp park depth, (<5% cover).

2.3.1 Prediction data layers

The availability of synoptic data on important factors driving kelp distributions around the UK permits the analysis presented here. The following sections give the provenance of this data.

2.3.2 Wave fetch



(a) (b)

Figure 2.5. (A) Offshore wave height around the UK and (B) inshore wave exposure derived from wave fetch (<5km from the coast). Offshore wave height data from the UK Atlas of Marine Renewables (ABP Marine Environmental Research Ltd 2008), and patterns of inshore wave fetch from a model described in Burrows (2012).

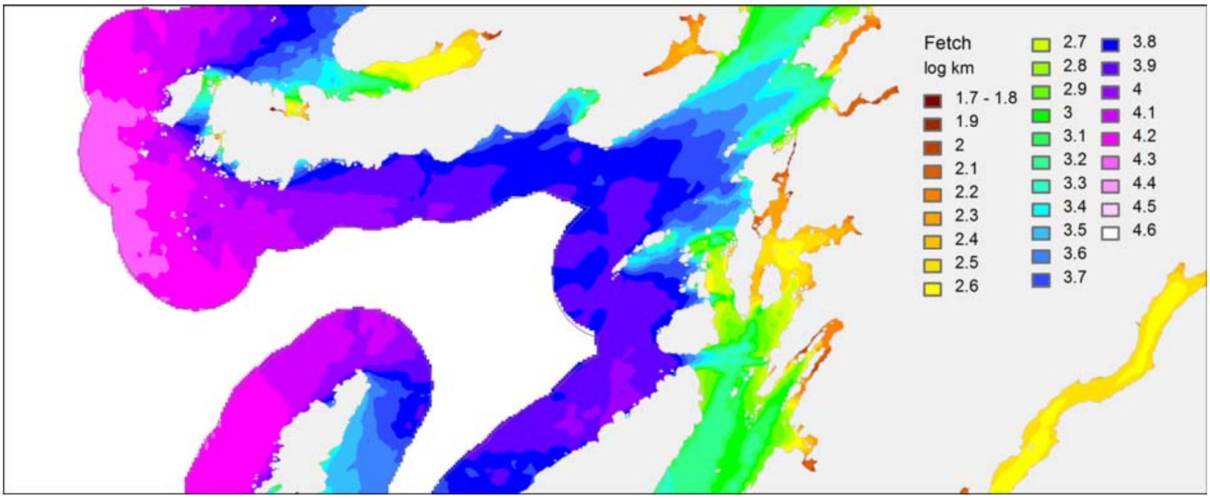


Figure 2.6. 200m-resolution wave fetch mapping of west Scotland, showing wave-exposed (purple) and wave-sheltered (yellow/orange/red) habitats. From Burrows (2012).

While relative wave exposure on a local scale can be expressed by a summative index of wave fetch (see below), regional differences in wave exposure may be more related to the oceanic wave climate than variation in wave fetch (Figure 2.5B). Wave height data from models show the general pattern of larger waves with greater proximity to the open Atlantic. Wave height is generally highest on open Atlantic-facing coasts, especially in the North and West of Scotland, west of Ireland and along the Western Isles and least along the coasts of the semi-enclosed Irish and North Seas and the English Channel (Figure 2.5A), with average wave heights off the Outer Hebrides in excess of 3m, but reaching only 1.2-1.5m through most of the English Channel.

Importantly, however, the influence of offshore oceanic waves is strongly modified by coastal topography. The local shelter offered by headlands and islands much reduces wave action, and very enclosed areas such as firths, channels and sea lochs have very little wave action (Figure 2.6). Wave fetch, the distance over which winds blow before reaching any piece of land, is a good predictor of wave height. Wind-generated waves follow well-understood physical laws, and wave heights and spectral characteristics (mixture of short-period and long-period waves) directly depend on the length of time that winds of specified velocities blow over the water surface. Even without including the more complex physics, simple indices of wave exposure have proved effective in predicting patterns in coastal ecosystems. Wave exposure indices range from relatively simple, such as a count of the number of sectors open to the sea (Baardseth 1970) to complex, using a sum of wave fetch values in all directions open to the sea, and weighted by the incidence and average speed of winds from those directions from local meteorological data (Thomas 1986). Fetch-only indices can perform as well as those including wind information, especially over areas where the wind pattern is relatively consistent (Burrows *et al* 2008).

Here we use the measure of wave fetch based on one produced during the UK MarClim project (Mieszkowska *et al* 2005; Burrows *et al* 2008) to establish the prevailing physical conditions at any survey site, and derive expectations for the composition of the community at that site. The index is based on the minimum distance (km) to the nearest land in each of 32 angular sectors of 11.25°, up to a maximum of 200km per sector (see Figure 2.7 for details of the method).

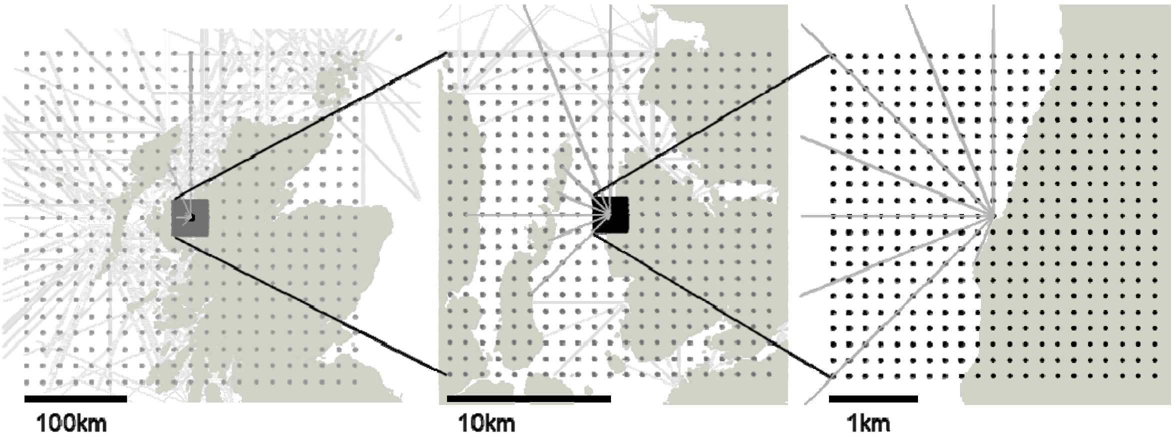


Figure 2.7. Determining wave fetch for a coastal location according to the method of Burrows *et al* (2008). Three spatial scales are searched for the nearest land in each specified direction: (left) every 20km up to 200km distant, (middle) every 2km up to 20km distant, and (right) every 200m up to 2km distant.

Given the wide range of values around the UK, from 1 to 32000km, it has been most convenient to express these as log base 10 values from 0 to 4.5 (Figure 2.6). The model has a spatial resolution of 200m. This spatial resolution matches that for MNCR location data, given as an Ordnance Survey grid reference to nearest 100m. The model is fine-scaled enough to capture relatively local-scale variation in wave exposure such as that found in the lee of headlands and small islands.

2.3.3 Temperature

Temperature is a key driver setting geographic patterns of marine species distribution (Hutchins 1947; Vermeij 1978; Lüning 1990), including kelp, although other proximate factors such as habitat type, extent and local hydrography can determine local abundance and be involved in setting range edges. Sea Surface Temperature (SST) derived from satellite images, is a widely available variable which can be used for biogeographic studies.

Maps of average patterns of SST: winter (February), spring (May), summer (August) and autumn (November) show the general pattern of increasing temperatures from the south to the north around the UK (Figure 2.8). The axis of change varies from summer to winter, with February SST generally higher in the west (Atlantic and Irish Sea) than in the east (North Sea), and August SST warmer in the southern areas of both east and west coasts of the UK. Regional scale patterns also vary between summer and winter. Summertime patterns show evidence of stratification, mostly delineated by the presence of tidal fronts (Pingree & Griffiths 1978), such as the Islay and Celtic Sea fronts, separating stratified areas with warmer surface water from tidally well-mixed areas with cooler surface water. Locally cooler areas such as the north-west tip of Cornwall may also influence the relative proportions of cold and warm water species, and are important to note.

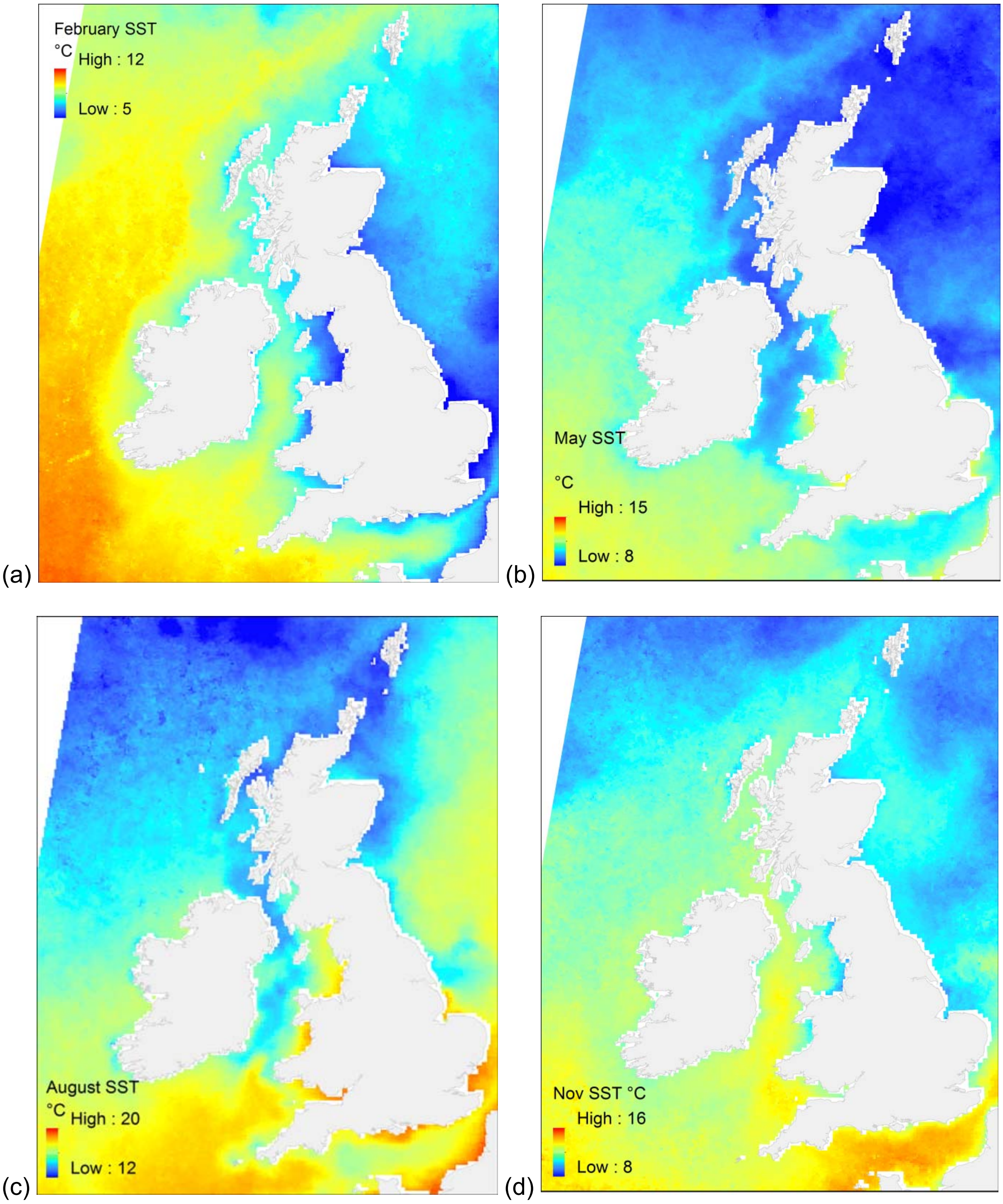


Figure 2.8. Sea surface temperature around the UK for (a) February, (b) May, (c) August, and (D) November; from averages over the period 2000-2006 obtained from the NASA Giovanni Data Portal (DAAC 2008).

2.3.4 Satellite chlorophyll *a* estimates as an indicator of light attenuation

Marine macroalgae including kelp are strongly influenced by large-scale patterns in water quality that result from spatial variation in the biomass of phytoplankton and the quantity of suspended material per unit volume. This mostly reflects the eutrophication status of inshore waters. Phytoplankton biomass is generally higher in areas of high nutrients (Cloern 2001). Data on contaminant concentrations are available, as in Charting Progress 2 (U.K. Marine Monitoring Assessment Strategy community 2010) for example, but not so widely or with such spatial resolution as satellite remote sensing. General patterns of contaminant concentrations follow the patterns of phytoplankton and sediment load, with highest values around population centres. Phytoplankton and sediment influence the spectral characteristics of reflected sunlight, but are not always easy to distinguish. The presence of significant amounts of suspended sediments makes estimation of the chlorophyll *a* concentration problematic. As a consequence, inshore coastal waters are considered differently from offshore oceanic waters (see, for example, Joint & Groom 2000). Estimates of chlorophyll in coastal waters (Case II waters) are increased by coloured dissolved organic matter and by suspended particulates from rivers or from the seabed. Notwithstanding such issues, the influences of phytoplankton biomass and suspended sediment on rocky shore biota are likely to be similar and for the purposes of developing indicators can be considered together. Increased phytoplankton and sediment increase the attenuation of light through the water, reducing the depth to which sufficient light penetrates to allow photosynthesis ('compensation depth' – where respiration of plants exceeds the photosynthesis). The euphotic depth is much reduced in areas of high sediment and phytoplankton, such as in the Bristol Channel, southern North Sea coasts and Liverpool Bay (Figure 2.9). The abundance of sublittoral macroalgae is much reduced in these areas (Burrows 2012).

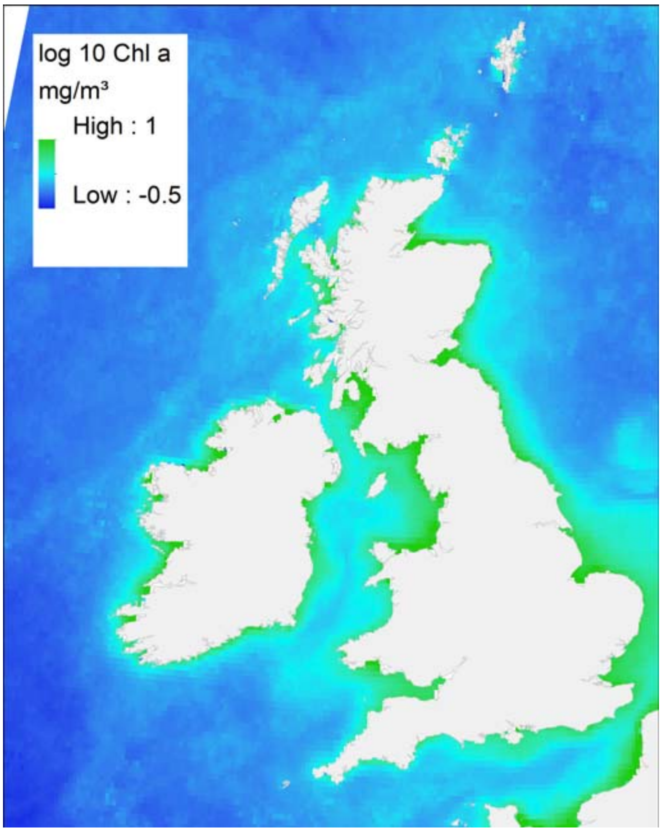


Figure 2.9. Chlorophyll *a* concentration ($\log_{10}$ mg/m^3) from MODIS-Aqua data averaged over every month from July 2002 to July 2012.

Here chlorophyll *a* concentrations are referred to as mg/m^3 in the knowledge that these values are influenced by suspended sediment, and that the probable effects on kelp species, habitats and communities are due to a complex set of interrelated processes, probably due to a combination of shading of macroalgae, greater retention of larvae of suspension feeders and enhanced feeding conditions for this same group (Burrows 2012). The data were obtained from the NASA MODIS Aqua satellite sensor, averaged over the period from July 2002 to July 2012. The coarser resolution of the NASA satellite is used rather than finer scale models such as the 4.5km light penetration model for UKSeaMap 2010 and the 1km resolution model used for EUSeaMap. The greater resolution of these latter models may appear advantageous, but may misrepresent the highly dynamic nature of phytoplankton concentrations at this scale. The data used are aimed at resolving differences on 10 to 100km scales in a primarily biogeographic or coastal cell context, and it is appropriate that for MSFD purposes we use data with similar resolution. Ultimately if the proposed indicators are adopted there will be the need to validate satellite derived information with data collected from inshore surveys. This is, however, beyond the scope of this report.

2.3.5 Other potential predictors, including depth

Other potentially useful predictors of kelp distribution and abundance may include tidal flow, nutrients, substratum availability and, particularly importantly, depth. The last of these, as a high resolution dataset at sub-100m scales, would be an invaluable resource especially when combined with information on substratum. Modelling of the distribution of *Laminaria hyperborea* in Norway has been successful in generating useful maps of kelp habitats on the same scales as these habitats have been sampled by MNCR (Bekkby *et al* 2009). Bekkby and her co-workers found that the terrain characteristics of a 25m-resolution digital elevation model were important in predicting kelp abundance. Rock surface slope, aspect and terrain curvature all contributed to the predictive power of the kelp model.

Available depth data include the GEBCO 1-minute grid, the SeaZone 250m DEM and BGS's DigBath250 bathymetric contour data (McBreen *et al* 2011), though restrictions on access to, and use of, the last two make it difficult to evaluate the potential for use in analyses. Since the MNCR data comes with depths recorded directly by the survey divers, these datasets would not improve the estimates of depth for the biological data. The low resolution of these data (GEBCO 1 minute is 1-2km) and the uncertainties and gaps associated with it (SeaZone data is not available for Scottish sea lochs, for example), render the use of these data in predictive modelling at the appropriate scale (200m) inappropriate without considerable further work. This lack of utility of depth information contrasts with the wave fetch data where modelled data does provide a better estimate of conditions than observations made on the day of the survey.

Tidal flow did have some influence in the study conducted by Burrows (2012) but was correlated with wave exposure. Rapid flows occur around headlands and islands in many places around the UK, but the best dataset available (ABP Marine Environmental Research Ltd 2008) for tidal flows based on a 2km tidal model from the Proudman Oceanographic Laboratory (now NOC Liverpool) does not resolve many of the rapid tidal channels in wave sheltered areas amongst the islands of the west coast of Scotland. For this reason, and because the data was not available on the appropriate spatial scale, tidal flow was not included in this analysis of kelp habitat distributions.

2.3.6 Covariation among environmental predictors

Environmental data for the different predictors of kelp habitat are correlated to a certain extent (Figure 2.10). This has an influence on the degree to which different predictors can be considered to have an independent effect on kelp abundance. In statistical terms this means

that the analysis is *confounded*. However, this is only really evident for chlorophyll *a* concentrations and February SST. Locally cooler areas in winter in the coastal eastern Irish Sea and in the southern North Sea have the highest average chlorophyll *a* concentrations, so a positive effect of February SST may be related to a negative effect of water-column chlorophyll *a* concentrations. Wave fetch varies on generally smaller scales, such that high and low wave exposure occurs in all other combinations of SST and productivity.
Despite this, SST (either February or August), wave fetch and chlorophyll *a* are used in statistical models of kelp distributions.

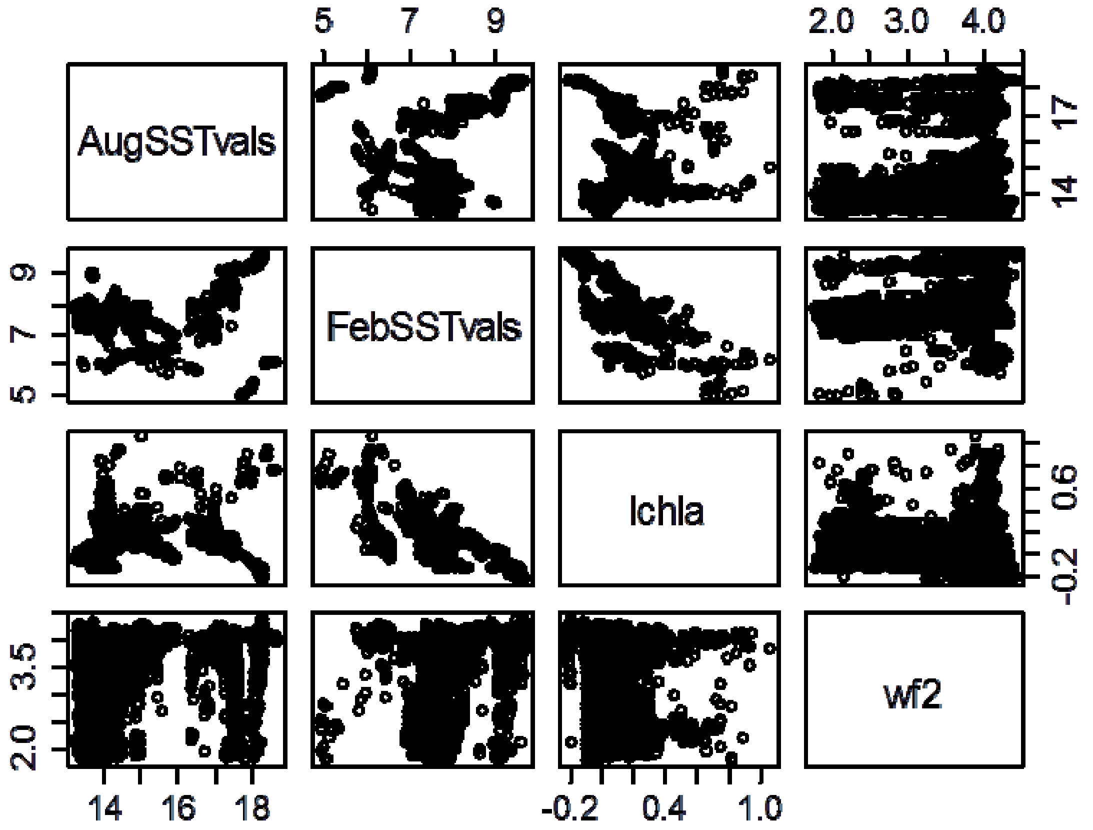


Figure 2.10. Covariation among environmental predictors of kelp distribution in the UK. Each point represents conditions at a single site surveyed by the Marine Nature Conservation Review. Abbreviations: AugSSTvals, August sea surface temperatures; FebSSTvals, February sea surface temperatures; lchla, chlorophyll *a* concentrations in $\log_{10}$ mg/m^3 units; wf2, summed wave fetch in $\log_{10}$ km.

2.3.7 Summary: final selection of environmental predictors

Summarising the considerations above, the final choice of predictors was as follows, and was determined by (i) applicability to kelp distributions, (ii) availability and (iii) ubiquity (across the UK):

Included in the analysis:

1) **Sea surface temperature** *(satellite derived, 9km resolution)*: this is the best predictor of large-scale geographical trends in kelp abundance, representing either predominantly north-south trends as for August SST or east-west trends as for February SST.
2) **Chlorophyll *a*** *(satellite derived, 9km resolution):* this best represents variation in light attenuation in the water column, resulting in the restriction of kelp to shallow depths, and can reflect coastal eutrophication.

3) **Wave fetch** *(modelled, 200m resolution):* large often delicate plants such as kelp are sensitive to water motion associated with waves and can be damaged by storms. Wave fetch data effectively predicts the distribution of robust and more sensitive kelp species.

Not included in the analysis:

4) **Depth:** current inshore bathymetry is not good enough at the scale of MNCR surveys to add information to the predictive models (although would help in visualising predictions).
5) **Tidal flow:** similar to depth, tidal information is not presently resolved at small enough spatial resolutions to apply to MNCR survey data locations. Tidal flow also generally (though not always) correlates with wave exposure.

2.4 Predictive models for kelp species

The suggested approach to developing indicators of Good Environmental Status presented here is to express the definition underpinning MSFD Descriptor 1 (*Biological diversity is maintained. The quality and occurrence of habitats and the distribution and abundance of species are in line with prevailing physiographic, geographic and climatic conditions)* in strictly quantitative terms, using the best available data collected using proven methodologies. That dataset is the data collected for NCC and JNCC under the Marine Nature Conservation review (MNCR) (see Section 2.2.1).

GES within this context would reflect that species are present as expected, given the presence of the kind of habitats where they normally occur, and the prevailing physical conditions. Statistical models that fit abundance data to environmental predictors allow objective analysis of patterns in respect to spatial gradients and temporal changes that can give indicators with a degree of supporting evidence.

The statistical approach used is proportional odds logistic regression, fitted in the R statistics package (version 2.15.0) using the `polr()` function. The `polr()` function fits a logistic regression model to an ordered factor response, in this case the ordered abundance categories on the SACFORN scale (Super-Abundant, Abundant, Common, Frequent, Occasional, Rare, Not Seen). Full details of the model fitting procedures are provided in Appendix 2. Models were selected on the basis of the best possible statistical fit to the data, tempered by the need for inter-comparison among the five species and the requirement that interaction terms in the models (the modification of the response to one factor by the level of another) be easily explainable in ecological terms.

Model parameters are summarised below for each species (Table 2.1). Logistic regressions deal with log odds ratios (using logarithms to base *e*), so parameter values indicate how more or less likely the outcome becomes as that predictor changes, given that model predictions are in probabilistic terms. For example, at 10m the log odds of achieving at least the same abundance score as at the surface for *Laminaria hyperborea* would be -10m x 0.121 = -1.21, giving $e^{-1.21}$, that is, 0.30 or 30% as likely as at the surface. Similarly, for each degree increase in August SST, the log-odds of *L. hyperborea* being at least ‘Common’ would decline by -0.314, that is the species would be $e^{-0.314}$ or only 73% as likely as at the 1-degree lower temperature. *Alaria esculenta* is more sensitive to increases in August SST: a 1-degree increase would reduce the likelihood of being at least ‘Abundant ‘ (or any class other than N “absent”) by $e^{-0.553}$ or 58% of that at the 1-degree lower temperature. Thus the sign and the magnitude of the parameter value gives the change in probability per unit change in the predictor to which it applies, expressed in log-odds terms.

Model predictions are given in two ways, (i) as the likelihood of a particular abundance class, or (ii) as the most likely class for that set of input predictors. The former is also additionally, and most usefully expressed as a cumulative value: the likelihood of abundance being *at least* a particular abundance class. Where that class is 'Rare' on the SACFORN scale, the model predicts the likelihood of the presence (and thereby absence) of that particular species. While the 'most likely class' can be the most easily understood form of model output, this may miss the nuances of predicted change.

Table 2.1. Summary of parameters relating kelp abundance to environmental predictors, from proportional odds logistic regression models fitted to MNCR abundance data. See Appendix 2 for full details of models and fitting procedures.

		Temperature (°C)			Wave fetch		Interaction
Kelp Species	Depth (- m)	August	February	Chl *a* ($\log_{10}$)	linear	quadratic	Depth x Chl *a*
Laminaria hyperborea	0.121	-0.314	-	-1.090	2.792	-0.252	0.154
Saccorhiza polyschides	0.117	-	0.489	-0.991	7.111	-1.044	0.033 [a]
Laminaria digitata	0.315	-	0.368	1.207 [a]	0.295	-	0.261 [a]
Saccharina latissima	0.127	-0.280	-	0.154 [a]	3.502	-0.641	0.150 [a]
Alaria esculenta	0.187	-0.553	-	-0.517 [a]	1.852	-	0.332

Note: values marked with superscript [a] have associated t-values of less than 1.96 and are unlikely to be statistically significant.

2.4.1 How can baseline predictions from models provide indicators?

Combining predictions from models with observations from surveys (see Section 3.2) allow indicators to be expressed in the form of model residuals**.** These residuals directly address Descriptor 1 of the MSFD. "*1. Biological diversity is maintained. The quality and occurrence of habitats and the distribution and abundance of species are in line with prevailing physiographic, geographic and climatic conditions*", particularly the latter part. Residuals show whether the observations exceed or fall short of the values expected from the environmental conditions at a particular location or across a collection of sites.

For continuous response variables, residuals can be inspected graphically (see for example Figure **2.7** in Burrows *et al* 2014), and the concept of exceeding a particular threshold of abundance or species richness is relatively easy to grasp. For ordinal response variables, residuals are harder to express. The Pearson residual is one measure of model residual that may be applied to ordinal data, although it only applies to binary data. The abundance class needs to be set in advance (that is, whether the kelp abundance data should be treated as exceeding or not exceeding a particular category). Pearson residuals are given as:

$$\frac{O_{ij} - E_{ij}}{\sqrt{n_i \hat{p}_{ij}\left(1 - \hat{p}_{ij}\right)}}$$

where O_{ij} and E_{ij} are observed and expected frequencies of (at least a particular) abundance class in a group of surveys (denoted by the letter i), n_i is the total frequency and $\hat{p}_{ij}$ is the probability of the abundance category. The deviance residual gives a similar measure of the difference between observed data and that expected from the regression model:

$$\pm 2\sqrt{O_{ij}\log\left(\frac{O_{ij}}{E_{ij}}\right)+\left(n_i-O_{ij}\right)\log\left(\frac{n_i-O_{ij}}{n_i-E_{ij}}\right)}$$

With data on kelp distributions in areas of known human pressures, it is possible to compare the observed frequency of abundance classes in the SACFOR scale with that expected from prevailing biogeographical trends, expressed by the ordinal logistic regression models. A good validation of the indicator would be if the residuals were negative in areas of human impacts and (slightly) positive or zero in otherwise non-impacted areas.

By mapping the distribution of these residuals (

Figure **2.11**) around the UK, it is possible to show areas where kelp is less abundant than expected. Interesting patterns emerge from this analysis for *Laminaria hyperborea* (

Figure **2.11**). The residuals are negative in the Firth of Clyde and positive on the west coast of Scotland. Similarly, residuals are negative around the mouth of the Tyne and Tees. If negative residuals can be shown to be associated with areas of increased human pressures, this metric may be used to indicate GES. Further development of the kelp species indicators would benefit from research into the pattern of model residuals around the UK using MNCR data.

More considerations of using the regression models as the bases of indicators are given in Section 2.5.

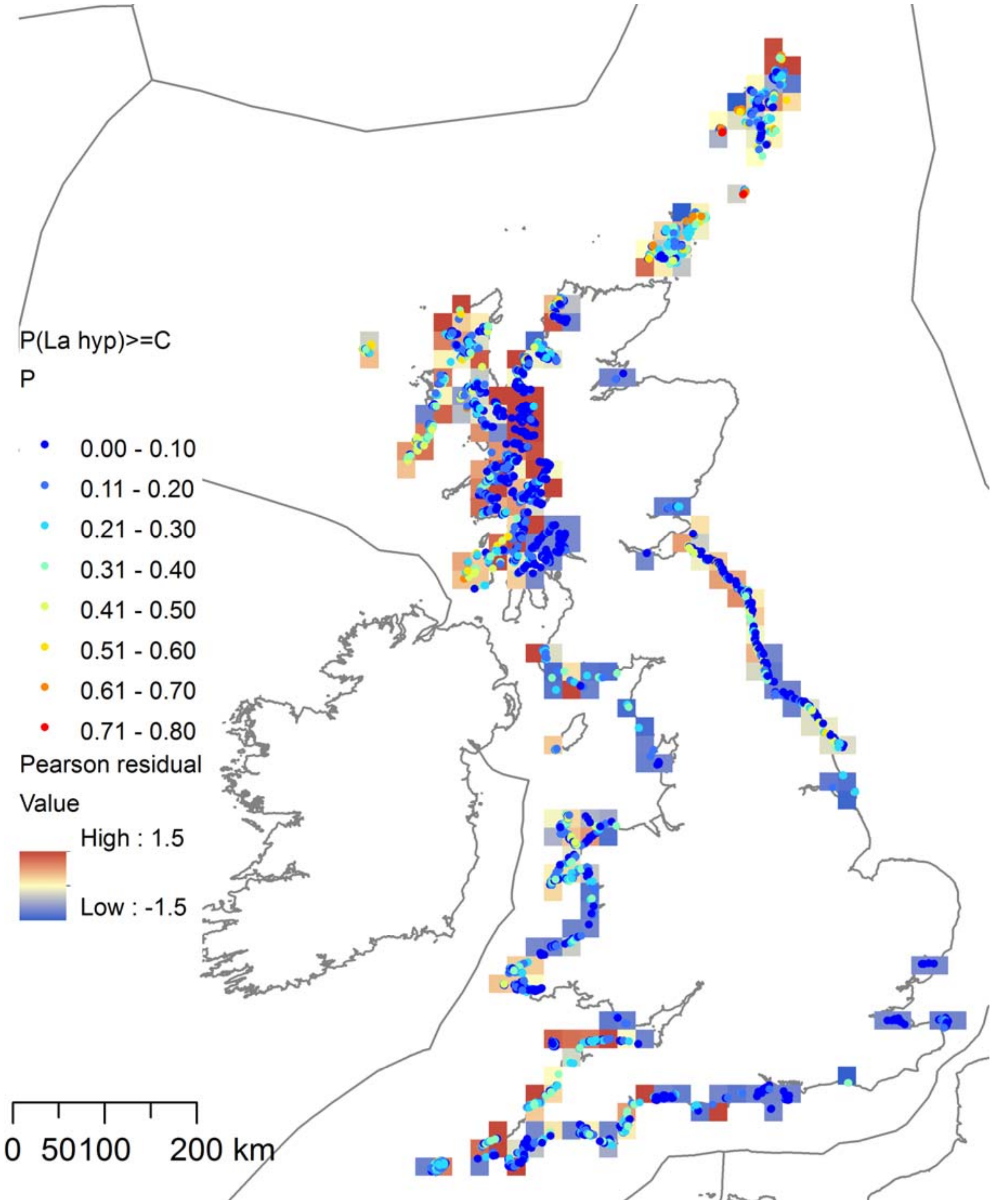


Figure 2.11. Pearson model residuals for the fitted *Laminaria hyperborea* model, calculated over 20km grid squares and using observed and expected frequencies of abundance greater than or equal to ‘Common’ (>10% cover). Positive residuals (red) show where kelp is more abundant than expected from the biogeographical trends fitted to MNCR data, and negative residuals (blue) where there is less kelp than expected. Circles give the expected probability of finding *L. hyperborea* as ‘Common’ or greater at each survey site.

2.4.2 Models by species

Visualizing the modelled distributions across four separate predictors (depth, temperature, chlorophyll *a* and wave fetch) is done here by a series of two-dimensional plots of the predicted probability of kelp species being at least 'Common' (>10% cover of the substratum) on the SACFOR scale. In each of these plots (Figure 2.12 to Figure 2.18), depth is presented on the vertical axis to aid conceptualisation of the patterns, and a second factor forms the horizontal axis. Figures for individual species distributions use wave exposure as the horizontal axis, with predictions given for four combinations of the remaining two predictors: low and high values of SST and chlorophyll *a* concentrations respectively.

Laminaria hyperborea (Figure 2.12) is more likely to be found as at least 'Common' in areas of low chlorophyll *a* and high wave exposure, and least likely in areas of high chlorophyll *a* and low wave exposure. The species' distribution is better predicted by the pattern of August SST (a general North-South trend) than February SST (generally East-West). As the UK kelp species with the highest incidence at MNCR survey sites, predicted probabilities of occurrence are higher than the other four species (compare ranges of likelihoods shown in Figure panel legends: *Laminaria hyperborea* plots scale from 0 to 0.8 for the likelihood of being at least 'Common').

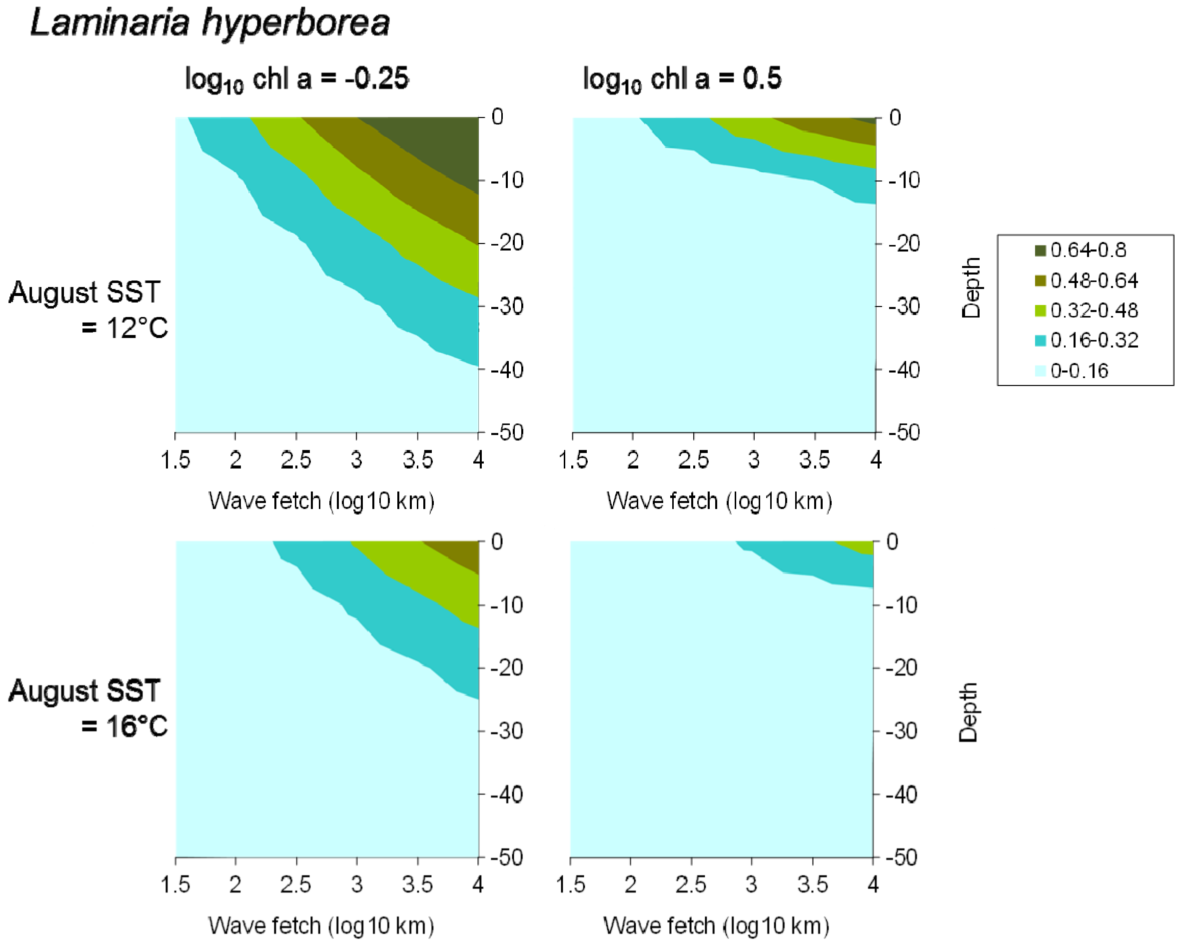


Figure 2.12. Probability of *Laminaria hyperborea* being at least 'Common' on the SACFORN abundance scale.

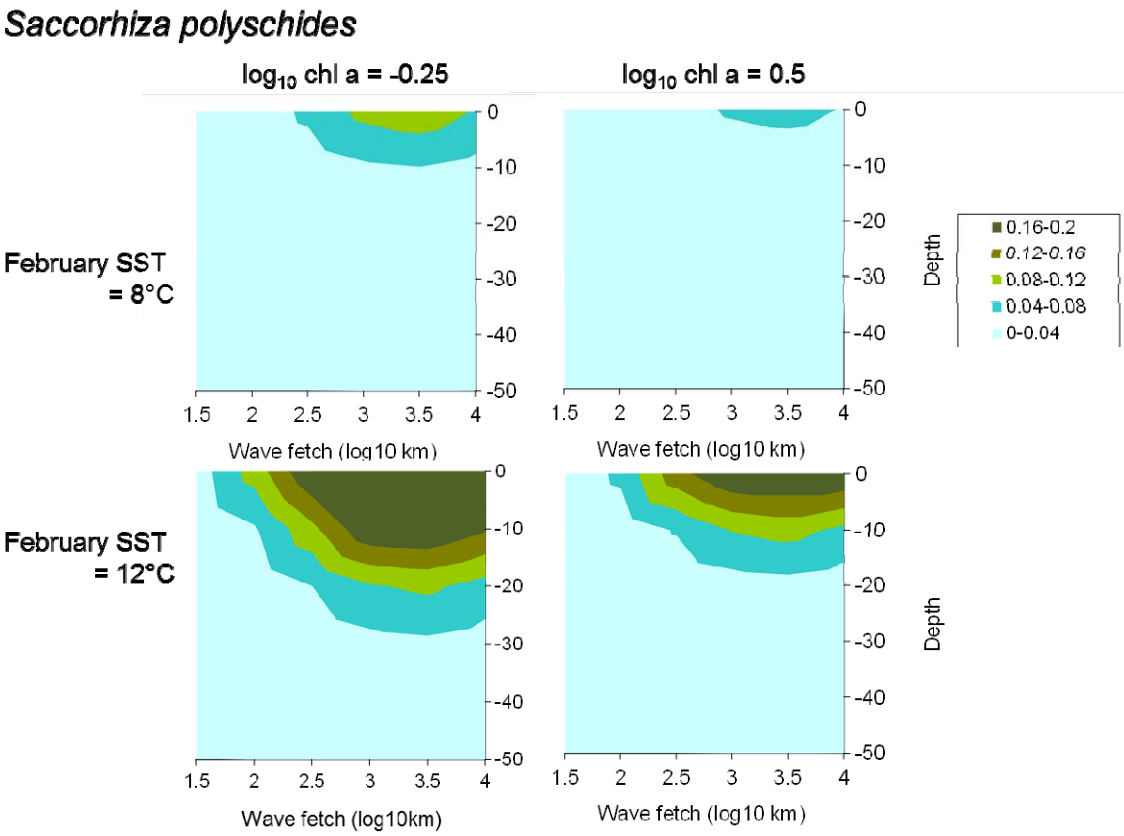


Figure 2.13. Probability of *Saccorhiza polyschides* being at least ‘Common’ on the SACFORN abundance scale.

Saccorhiza polyschides, as anticipated, is predicted by the fitted model to be most abundant in mid to high levels of wave exposure and much less abundant in low wave exposure (Figure 2.13), and not simply linearly increasing with wave fetch. Chlorophyll *a* did not have as negative influence as for other species (Table 2.1), suggesting that the species may be able to persist in areas of higher chlorophyll than the other UK kelp species. Once other factors are taken into account, the abundance of *Saccorhiza polyschides* increases with increasing winter SST.

The distribution pattern of *Laminaria digitata* across the range of the environmental predictors as shown by these models is most strongly influenced by its rapid decline with water depth (Figure 2.14). In contrast to the other four kelp species, it is relatively insensitive to wave action, and occurs across a wide range of values of wave fetch, and reflected in the low value for the linear term for wave fetch in the logistic regression model (Table 2.1). The species is predicted to be more likely to be common in areas with warmer winter SST.

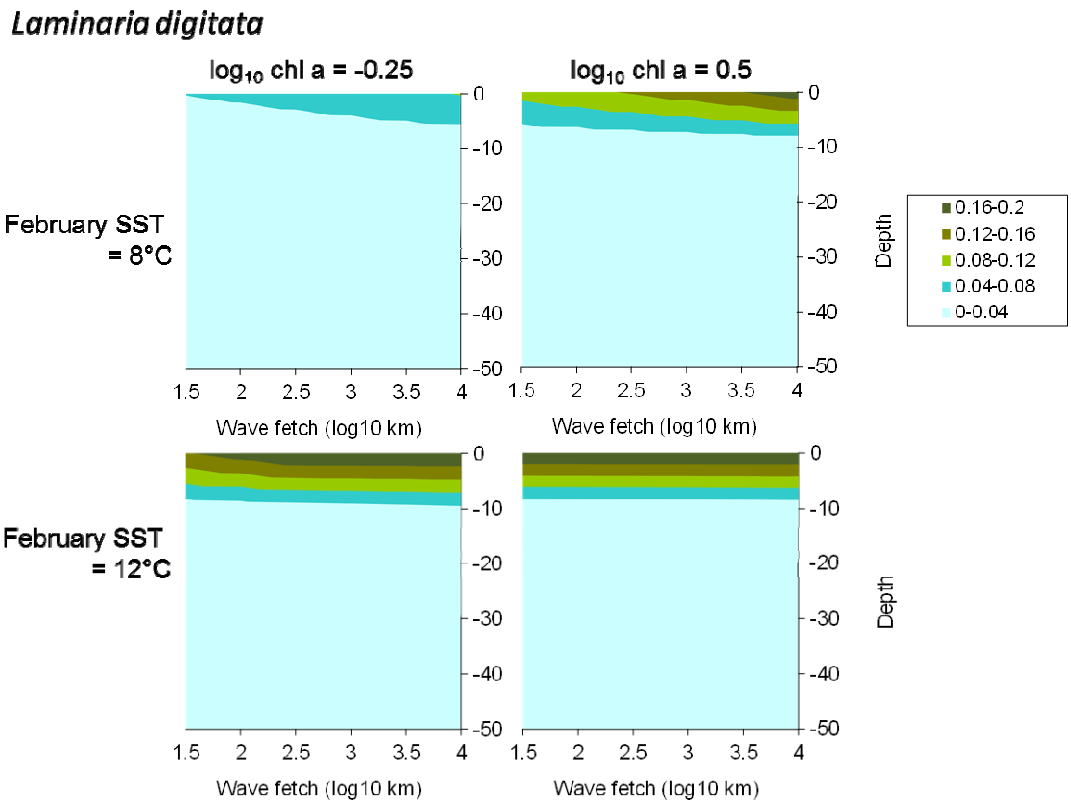


Figure 2.14. Probability of *Laminaria digitata* being at least ‘Common’ on the SACFORN abundance scale.

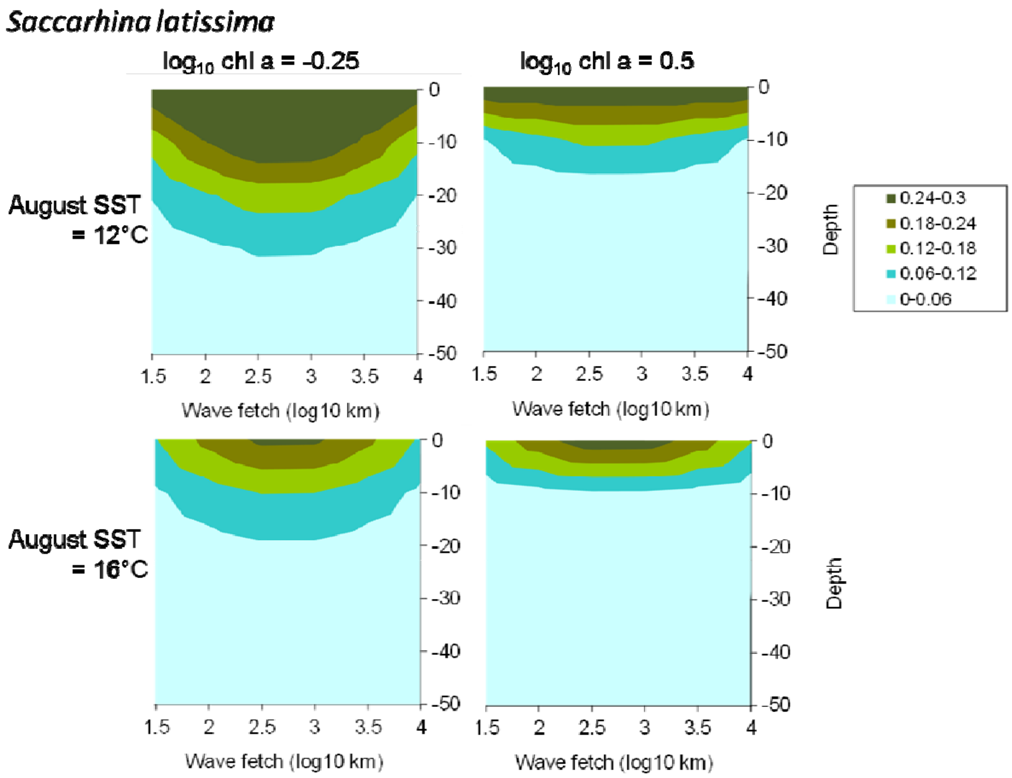


Figure 2.15. Probability of *Saccharina latissima* being at least ‘Common’ on the SACFORN abundance scale.

Sugar kelp, *Saccharina latissima*, is most likely to be 'Common' in areas of intermediate wave exposure (2.5 to 3.0), as shown by the significant quadratic term in the model (Table 2.1). The species is more abundant in areas of lower summer SST, and lower chlorophyll *a* concentrations.

Alaria esculenta is only prevalent in the most wave exposed conditions, and more abundant in the areas of cooler summer temperatures (Figure 2.16). The species is quite sensitive to the effects of higher chlorophyll *a* concentrations (shading) and is restricted to much shallower depths in those areas.

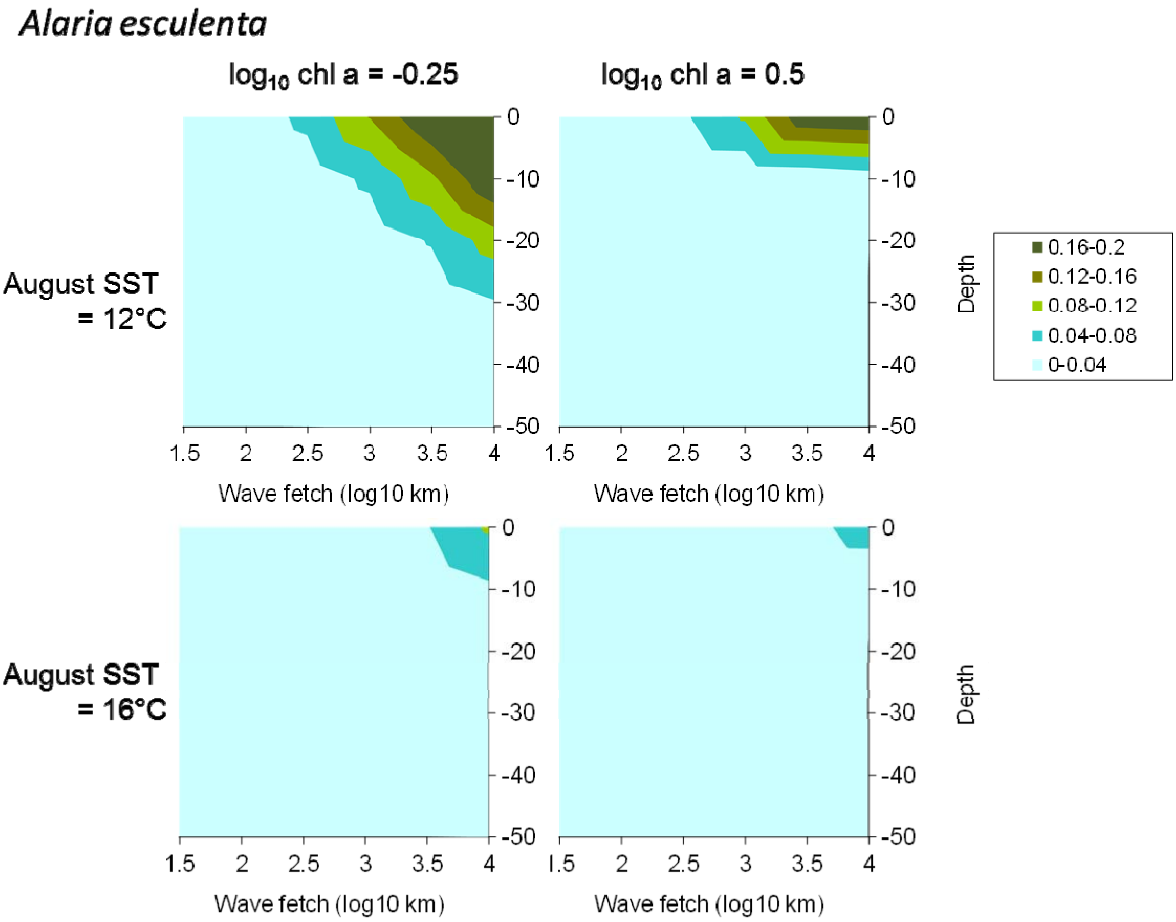


Figure 2.16. Probability of *Alaria esculenta* being at least 'Common' on the SACFORN abundance scale.

Of the five kelp species, three were more likely to be recorded as 'Common' in cooler August SST (*Laminaria hyperborea*, *Saccharina latissima* and *Alaria esculenta*), while the other two (*Saccorhiza polyschides* and *Laminaria digitata*) were more likely to be 'Common' in warmer February SST. This largely matches the previously recorded warm-water affinity of *Saccorhiza polyschides* in UK waters (see Chapter 1), and the cold-water preferences of *Laminaria hyperborea* and *Alaria esculenta*. The change in predicted abundance of *Saccorhiza* across the gradient of February SST is particularly marked.

Temperature responses were only modelled as a linear response, and an additional quadratic term for temperature may show more marked discontinuities in distributions across thermal gradients. Extra model terms do, however, increase the risk of model 'overfitting', in which additional parameters increase the fit of the model but only reproduce additional 'noise' rather than the signal provided by the response to the primary environmental drivers.

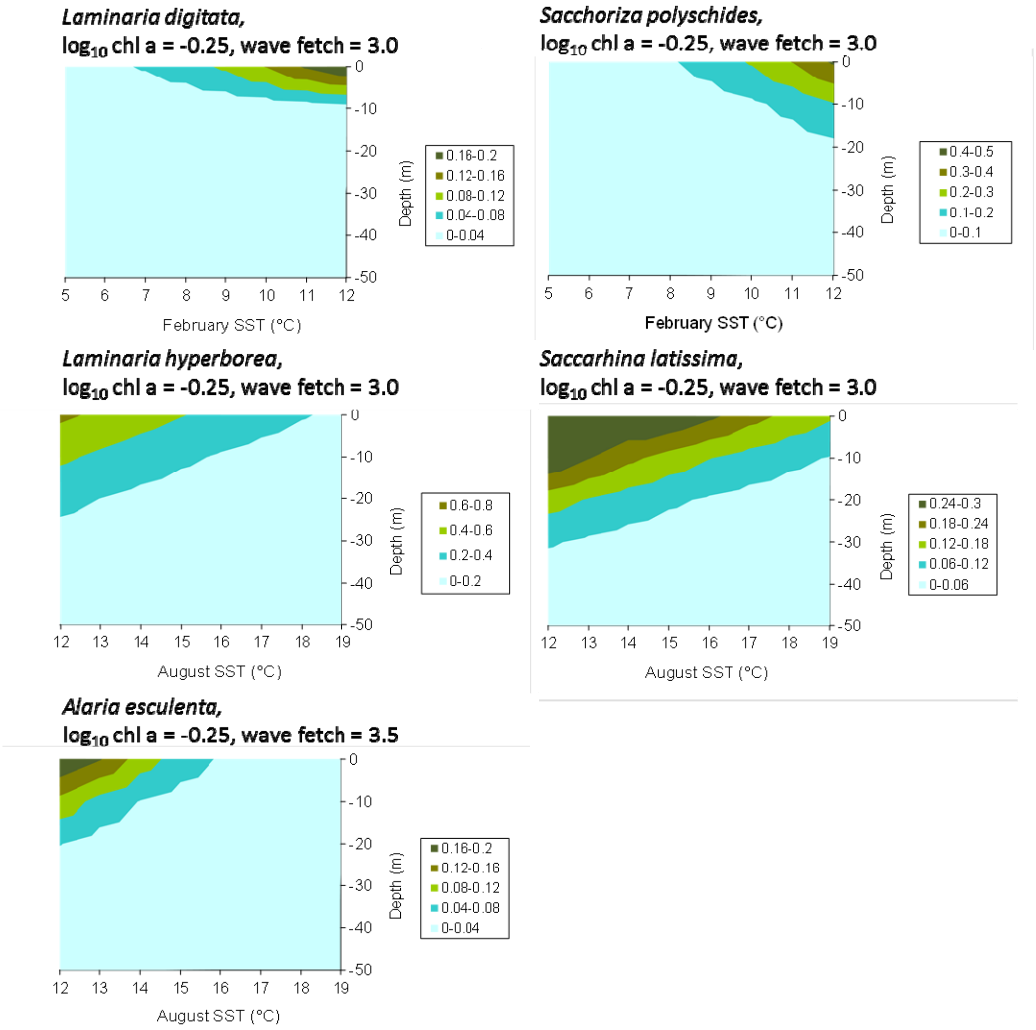
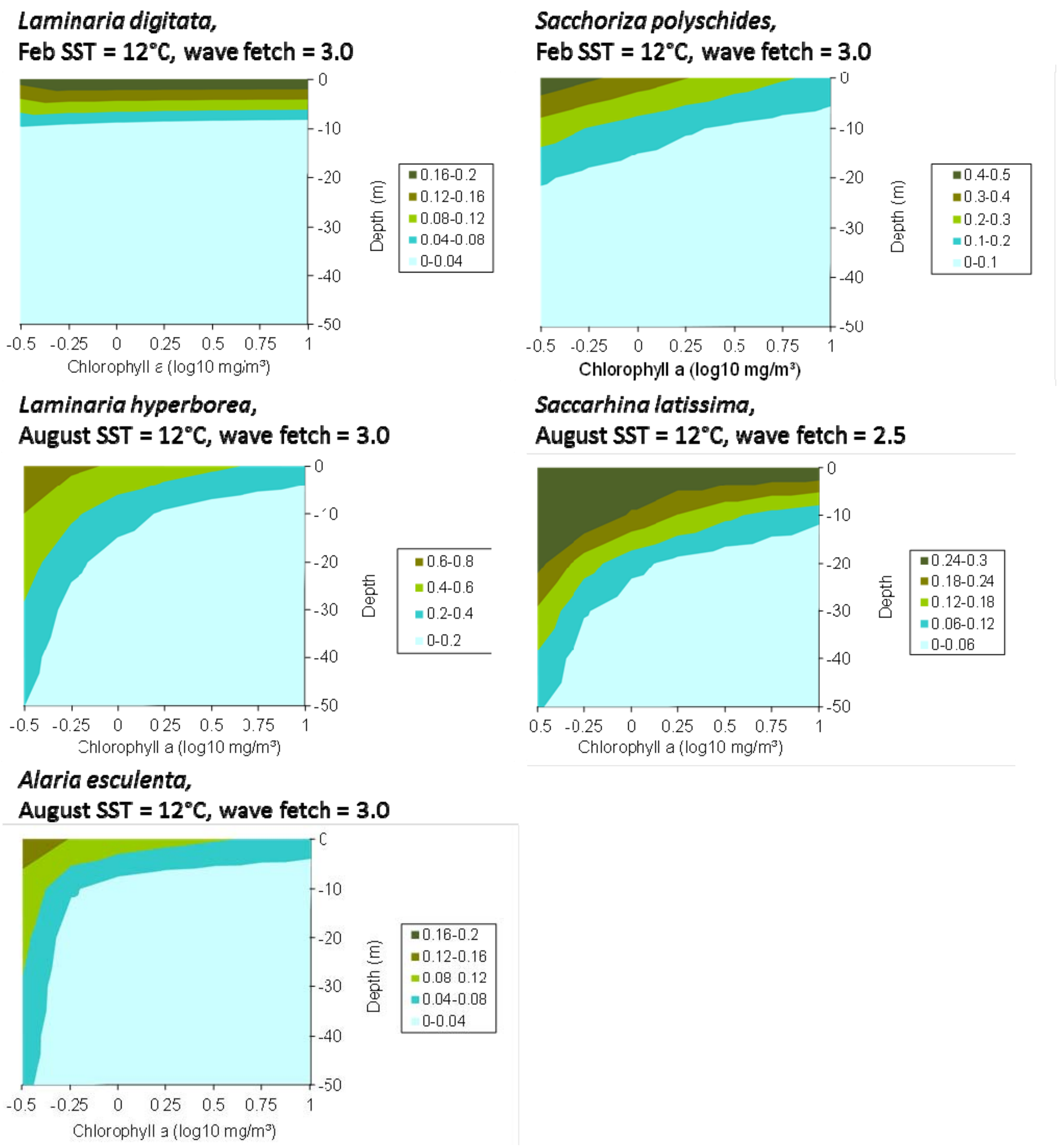


Figure 2.17. Temperature sensitivity of UK kelp species, show by changes in the predicted probability of being at least ‘Common’ on the SACFORN abundance scale.



Laminaria digitata, Feb SST = 12°C, wave fetch = 3.0

Sacchoriza polyschides, Feb SST = 12°C, wave fetch = 3.0

Laminaria hyperborea, August SST = 12°C, wave fetch = 3.0

Saccarhina latissima, August SST = 12°C, wave fetch = 2.5

Alaria esculenta, August SST = 12°C, wave fetch = 3.0

Figure 2.18. Sensitivity of UK kelp species to changes in chlorophyll *a* concentrations, show by changes in the predicted probability of being at least 'Common' on the SACFORN abundance scale.

Four of the five UK kelp species are negatively affected by increasing water-column chlorophyll *a* concentrations (Figure 2.18). *Laminaria hyperborea* and *Alaria esculenta* have responses to chlorophyll *a* that are strongly modified by the depth considered, with the effect of chlorophyll *a* increasing at greater depths (as indicated by greater magnitudes of the depth by chlorophyll interaction term in the regression model, Table 2.1).

2.4.3 Predictive habitat maps by species

A highly desirable and very useful outcome of this analysis is the ability to use these fitted models to predict and plot habitat maps by species. However, depth, the parameter to which kelp is most sensitive, changes rapidly over very small spatial scales. Furthermore, data on inshore bathymetry in areas of less than 30m depth, along with the maps of the incidence of rocky substratum, are not readily available. Given the relatively high spatial resolution of the wave fetch data (200m), once detailed bathymetry and maps of substratum types of shallow areas become accessible, these maps can be produced. Indeed, such data can be used to refine models further, since surface terrain measures (slope, aspect etc) can be used as additional predictors in models.

2.5 Using predicted presence and abundance for assessment of Good Environmental Status

In this report quantitative models have been produced that predict the abundance of kelp species that can be expected from both local habitat conditions, in the form of wave exposure, and from regional and biogeographical environmental variations, as seasonal average sea surface temperature (SST), pelagic primary productivity suggested by chlorophyll *a* concentrations, and/or the amount of suspended sediment in the overlying water column. These models offer probabilistic expectations for the well-known SACFOR system of marine data recording of abundance categories. This probabilistic form means that site- and depth-specific predictions of kelp abundance have limited use as the basis of an indicator of the expected status of kelp communities under the concept of "*in line with prevailing […] conditions*" of MSFD Descriptor 1. More usefully, the predictions can be aggregated for groups of sites to give the expected distribution of SACFOR scores in a regional survey, and divergence of observed abundance from this predicted distribution can indicate a potentially impacted kelp community (see Section 2.4.1,

Figure **2.11**).

The logistic regression modelling approach achieves two out of the four stated goals (Section 2.3) of predicting (i) the likely presence of each of the five major UK kelp species at a particular site by giving a likelihood for the recording of kelp species as at least 'Rare'; and (ii) the likely abundance of those species as a function of depth at a particular site, since the models predict the likelihood of each SACFOR category with depth they permit the estimation of the most likely category of abundance at each depth. The potential measures of kelp habitat that indicate Good Environmental Status (GES) are *kelp depth and kelp park depth.* The other two stated goals: (iii) the maximum kelp depth, below which kelp abundance is less than 20% cover; and (iv) the maximum kelp park depth, (<5% cover), are less directly linkable to the probability distribution predicted by the regression models. Instead of a single value for percentage cover, the models give a range of probabilities for different values of percentage cover. It would be possible to produce a single cover value by calculation of the average cover for each abundance category weighted by the probability of each category, but this would misrepresent the essential uncertainty captured by the modelling approach and furthermore the effect of the predicted likely absence of the species.

Given these considerations, uncertainty and the issue of the influence of predicted absence, two approaches present themselves for deriving the depths expected for cover >20% (kelp) and >5% (kelp park) at each locality. One approach would be to set threshold probabilities for the expectation of percentage cover of kelp exceeding 5% ('Frequent' on the SACFORN scale) and 20% ('Abundant'), either for probabilities combined across all species or for each one singly. Once threshold probabilities were set, regression model equations can be re-arranged to provide the predicted depth. A second approach would use the probability distribution of abundance categories to derive the weighted average percentage cover of kelp, and thereby the depth at which this weighted average value exceeded 5% and 20% respectively. The latter method would not predict any meaningful depths in regions where kelp incidence was low. A potential modification would be to calculate depths where 5% and 20% cover was exceeded contingent on kelp being present. This would be achieved by omitting the probability of the 'Not Seen (N)' category of the SACFORN scale from the calculation of weighted average abundance. Alternatively, a new set of abundance regression models could be fitted to data where the kelp species were recorded as present. The utility of these approaches remains to be explored.

Finally, a single index of GES for kelp habitat can be produced by aggregating achieved abundance scores across multiple species at a site and comparing these with the scores predicted by the models (the abundance expected as per Descriptor 1). By reducing the outcomes of the model to binary eventualities (i.e. is or is not greater than 5% or 20% cover, or ‘Common’ and ‘Abundant’ respectively), then the degree to which the kelp abundance in an area exceeds expectations can be quantified (see Section 2.4.1,

Figure **2.11**). Once thresholds are set for the achievement of GES on the basis of the abundance relative to the expectations based on biogeographical patterns, the index will be complete.

3 Research and development requirements for indicators of GES for kelp

In developing indicators of Good Environmental Status (GES) for sublittoral kelp habitats, the current state of knowledge of the ecology of kelp in UK waters has been reviewed (Section 1), and subsequently existing extensive data on distributions have been used to produce quantitative models of the expected condition of populations of kelp species around the UK (Section 2) as a foundation for assessment of GES. In this Section, a recommended approach to data gathering and outstanding research is outlined in order to enable the proposed indicator(s) to be made operational, and as requested an estimate of the likely cost of any extra research is also provided.

3.1 Knowledge gaps and research requirements for kelp habitats and species in the UK

In Section 1, the current state of knowledge and understanding of the dynamics of UK kelp populations and habitats was reviewed. In this Section the critical deficiencies in this understanding are highlighted, and research identified that is needed to address these issues, and which can enhance any judgement of the GES of kelp habitats.

3.1.1 Temporal variation in kelp abundance

A key unknown for UK kelp communities is the degree of inter-annual and longer-term variation in the abundance and biomass of kelp species. Very early on, Walker (1956a, 1956b) noticed considerable variation in annual average yields of kelp around Scotland in grab surveys, ranging from 7.2kg/m^2 in 1947 to 1.5kg/m^2 in 1953. Walker correlated the inter-annual changes with sunspot numbers; higher numbers usually being associated with warmer temperatures (see also Southward *et al* 1988). While the mechanism behind this large variation in less than a decade was not made clear, it demonstrates the importance of appreciation of the natural inter-annual variability in kelp. Good candidates for the reasons behind such variation include disturbance during winter storms having a negative effect, and clear sunny weather during summer enhancing production and leading to greater abundance and biomass in the same or the following year's surveys. These associations would suggest that the North Atlantic Oscillation (NAO) would be a good predictor for inter-annual changes in UK kelp populations.

Changes in the frequency, intensity and timing of storms may be affecting UK kelp populations and their subsequent recovery from storm disturbance. There is increasing evidence to suggest that changes in the Earth's climate system are altering weather patterns in the UK. For example, changes in the position of the jet stream, perhaps in response to changes in Arctic sea-ice extent (Hofer *et al* 2012) have resulted in changes to the timing, frequency and intensity of storm tracks across the Atlantic Ocean. The jet stream was much further south than usual during the summers of 2011 and 2012 resulting in a greater number and intensity of summer storms and a particularly wet summer, leading to increased freshwater run-off and resultant high levels of turbidity. Moreover, the more southerly latitude of the jet stream over the winter of 2013/14 led to an increased number and intensity of large winter storms leading to large amounts of kelp detritus and therefore disturbance of kelp beds in south-western England (Smale, pers. obs.), Wales (Moore, pers. obs.) and Northern Ireland (O'Connor, pers. obs.). While climate scientists predict that extreme events are likely to increase into the future, how changes in the timing, frequency and intensity of such events may alter kelp abundance, productivity and subsequent recovery is broadly unknown.

Elsewhere, there is strong evidence that climate fluctuations drive kelp abundance. A particularly well studied system is the Californian giant kelp *Macrocystis pyrifera* where kelp declines are associated with El Niño events (Dayton *et al* 1999). In that system, cold nutrient-rich La Niña periods alternate with warmer nutrient-poor El Niño periods, with lasting effects on the dominance of the main kelp species (Dayton *et al* 1999). Whether similar temporal changes are important in UK kelp systems is unknown. The strength of the evidence for climate-related change from the US kelp studies is due to the duration of the work, and the regularity of assessment of the system. A good model for a programme of coastal research is the US system of Long Term Ecological Research (LTER) sites. The Santa Barbara Coastal LTER project aims to determine the processes structuring Californian giant kelp forest ecosystems. Kelp is recognised as a key part of the coastal system in California, and is a major attraction at public aquaria, like that at Monterey Bay Aquarium (Figure 3.1).

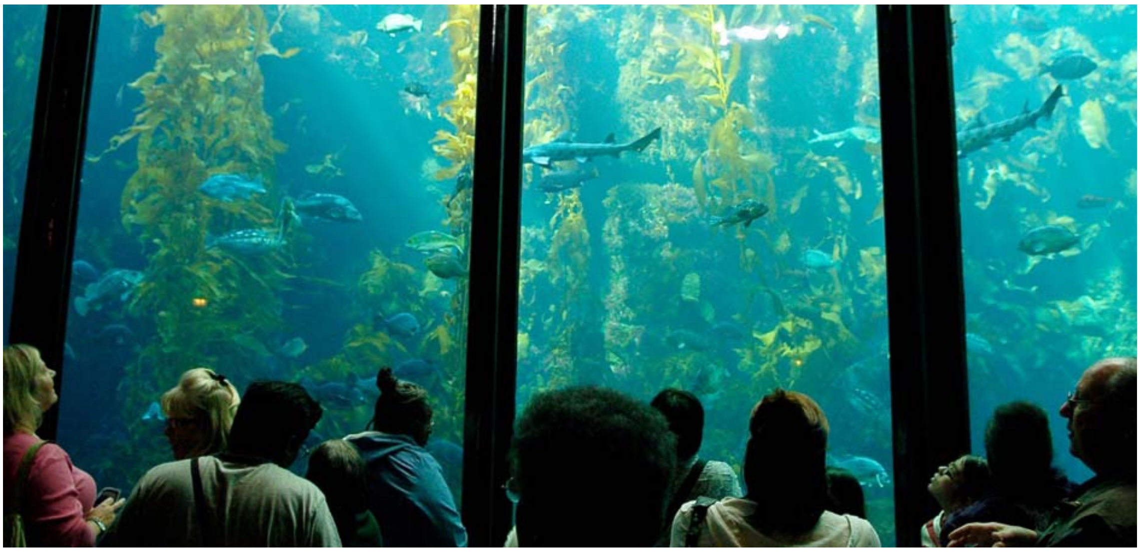


Figure 3.1. *Macrocystis pyrifera* and associated fauna on display at Monterey bay Aquarium (image from https://www.montereybayaquarium.org/animals-and-experiences/exhibits/kelp-forest)

3.1.2 Resilience, stability and food web structure

While the resilience, stability and food-web structure of kelp forests from many places around the world have been well studied and have indeed been instrumental in understanding ecological concepts such as trophic cascades and alternative stable states (Steneck & Johnson 2013), the understanding of these processes for UK kelp systems is rudimentary (Smale *et al* 2013a). Comparisons between the UK and better studied regions of the north-east Atlantic (e.g. Norway and Portugal) may be possible in order to provide insights into the effects of press (chronic) and pulse (temporary) disturbances. For example, work in Norway suggests that kelp forests are able to recover from disturbance within 7 years of the disturbance occurring. Care does, however, need to be taken with regard to how far these inferences can be made. Kelp beds are subjected to grazing, sometimes reverting to urchin barrens. Herbivory by urchins in Norway can lead to such barrens (Norderhaug & Christie 2009). Urchin barrens are considered rare in the UK, which may be a function of the increased diversity of kelp species in the UK compared to Norway leading to the more diffuse influence of herbivory (see Byrnes *et al* 2006, for an example of increased diversity leading to more diffuse top-down effects).

Kelp detritus underpins the base of coastal food webs, which include many commercially important species (e.g. *Nephrops sp., Cancer sp.*) and associated ecosystem services. Not much is known of this detritus production in the UK, though globally, the majority of kelp production enters detrital food webs (e.g. Krumhansl & Scheibling 2012b). Provisioning of detrital food webs is therefore a fundamental role of kelp forests. Estimating standing stock of kelp beds and relating this to estimated rates of ecosystem functioning is complicated by the inherent variability of kelp production at local spatial scales. Epibiotic loading, dislodgement following storms (predicted to increase under current IPCC prediction, (Byrnes *et al* 2011), climate induced species ranges shifts (important as the assemblages associated different species of kelp differ (Blight & Thompson 2008) and potentially diseases (e.g. *Paramoeba invadens*) may all have significant impacts on the stability and resilience of kelp populations. Conceptual models of kelp detrital production are being developed (e.g. Krumhansl *et al* 2014) but the paucity of baseline data for UK waters limits current models for this region. The Marine Ecosystem Research Programme (MERP) (http://www.marine-ecosystems.org.uk/) may help to address the fate of kelp-derived organic material in food webs. It is important that future kelp monitoring is conducted at a scale that will be most meaningful ecologically and can be used to develop models to predict kelp production and detrital input. There are many metrics to estimate community stability (Donohue *et al* 2013) and fixed monitoring of kelp beds will permit the estimation of several of these metrics devised to quantify resistance to and resilience following inevitable perturbation events (e.g. storms). In addition, the community data described in the proposed sampling protocol will allow us to produce space visualisations of kelp community states, based on estimate Euclidian distances to track change and variability, following the holistic methodological approach of the MSFD (Tett *et al* 2013). Seasonal sampling will also allow for temporal variation in kelp-derived organic matter supply to be incorporated into future models (Leclerc *et al* 2013; Rodriguez *et al* 2013).

3.1.3 Pressure-state-response information

Given that the ultimate use for indicators of GES under the MSFD is to gauge the response of the ecosystem to human pressures, it is essential that we understand how these indicators change in relation to such pressures. In the UK there are very few studies of the effects of human pressures on kelp. A notable exception is the study by Jones (1971, 1973) of the effects of industrial pollution on macro-invertebrate communities associated with kelp in north-eastern England. Jones (1973) noted changes in community composition along a complex pattern of coastal conditions heading from relatively clear water in the furthest north to naturally turbid in the south of the study area. Many species declined in the areas around the mouths of the rivers Tyne, Wear and Tees where high levels of organic and inorganic pollutants had been found. For a large group of species, including the limpet *Tectura testudinalis,* the gastropod *Rissoa* sp., and the sea slug *Aplysia punctata,* distributions along the coast did not include the most polluted open coastal areas, leading Jones to conclude that these were pollution-intolerant species. The pattern of residuals from the UK *Laminaria hyperborea* model fitted to MNCR data (

Figure **2.11**) also shows less-than-expected abundance of kelp in this area.

The coincidence of the polluted areas with increasing levels of suspended sediment, albeit some man made in the 1970s when mining waste and power station fly ash (500,000t per annum) was routinely dumped in the sea, makes it difficult to separate these two independent effects. In a similar study of kelp holdfast communities from the same area at approximately the same time, however, Moore (1973) concluded that separating polluted from unpolluted turbid sites was not tenable on the basis of a more rigorous statistical hierarchical clustering approach. Moore urged caution on the basis that correlation does not imply causation, an approach that it would be wise to adopt in attributing changes in faunal composition of kelp communities to pollution without further evidence. Stronger evidence linking declining kelp macro-invertebrates with increasing pollution is needed before recommending the greater taxonomic effort required to quantify such changes. More

research in this area may yet reveal species that are unequivocal indicators of pollution pressures.
As stated in Section 2.3.4, satellite-based estimates of the concentration of chlorophyll *a*, indicating abundance and biomass of phytoplankton, can be assumed to be a proxy for eutrophication status, though this may not all be human activity related. Decline in kelp populations along gradients of estimated satellite chlorophyll *a*, which also includes the effects of additional turbidity, is captured by the logistic regression models of the MNCR data (Section 2.4). Additional decline in kelp in areas of already high chlorophyll *a*/turbidity may suggest local eutrophication effects.

Evidence of pollution, particularly sewage, on kelp communities is stronger from other continents. The decline and recovery of the Point Loma giant kelp bed off Los Angeles has been considered to reflect increasing pressure from sewage discharges and subsequent improvement, played out against background of a rapidly fluctuating climate with episodic El Niño events (Foster & Schiel 2010). Indeed, the prevailing climate at the time can have a direct impact on the outcome of acute pollution events: the recovery of a giant kelp bed from a sewage spill was helped by the fact it took place in winter when waves could disperse the pollution and little growth of plants was happening (Tegner *et al* 1995). The *Macrocystis pyrifera* kelp recovered from this 1992 spill in San Diego quickly with little evidence for long term effects.

3.1.4 Spatial variability and physical mapping data

A valuable resource for understanding the likely status of kelp populations around the UK would be detailed bathymetric and habitat maps of the shallow sublittoral zone (<50m depth). There is presently a poor level of mapping at these depths, generally covering the white zone between Ordnance Survey and Hydrographic Office charts. One problem and possible cause is the difficulty in getting ships into such shallow water for acoustic (multibeam) bathymetric sensing to work properly.

Much effort is being expended at present to improve coastal mapping. The MAREMAP project (http://www.maremap.ac.uk/) is a joint initiative led by the British Geological Survey (BGS), the National Oceanography Centre (NOC) and the Scottish Association for Marine Science (SAMS). Funded by the UK Natural Environment Research Council (NERC), the project is producing highly detailed maps that would be extremely useful for predicting where kelp habitats should be, and for refining models on the basis of existing data (deriving slope and elevation characteristics for example) and for locating new surveys. The coverage resulting from the project illustrates the problem well (Figure 3.2).

Bathymetric LIDAR, a laser based system that depends on light penetration in shallow coastal waters (see for example http://www.fugro-pelagos.com/lidar/tech/lidar_bathy.html; http://www.fugro-pelagos.com/lidar/lib/brochures/Marine_4pg.pdf) can provide depth information in the shallow sublittoral. Swath bathymetry can also be extended into shallow water, an approach used for the Channel Coastal Observatory (http://www.channelcoast.org/) for surveys of the coasts of south-west England. The links between these physical data and past and future ecological surveys need to be made much stronger, and increasing availability of physical data for research will help considerably in this regard.

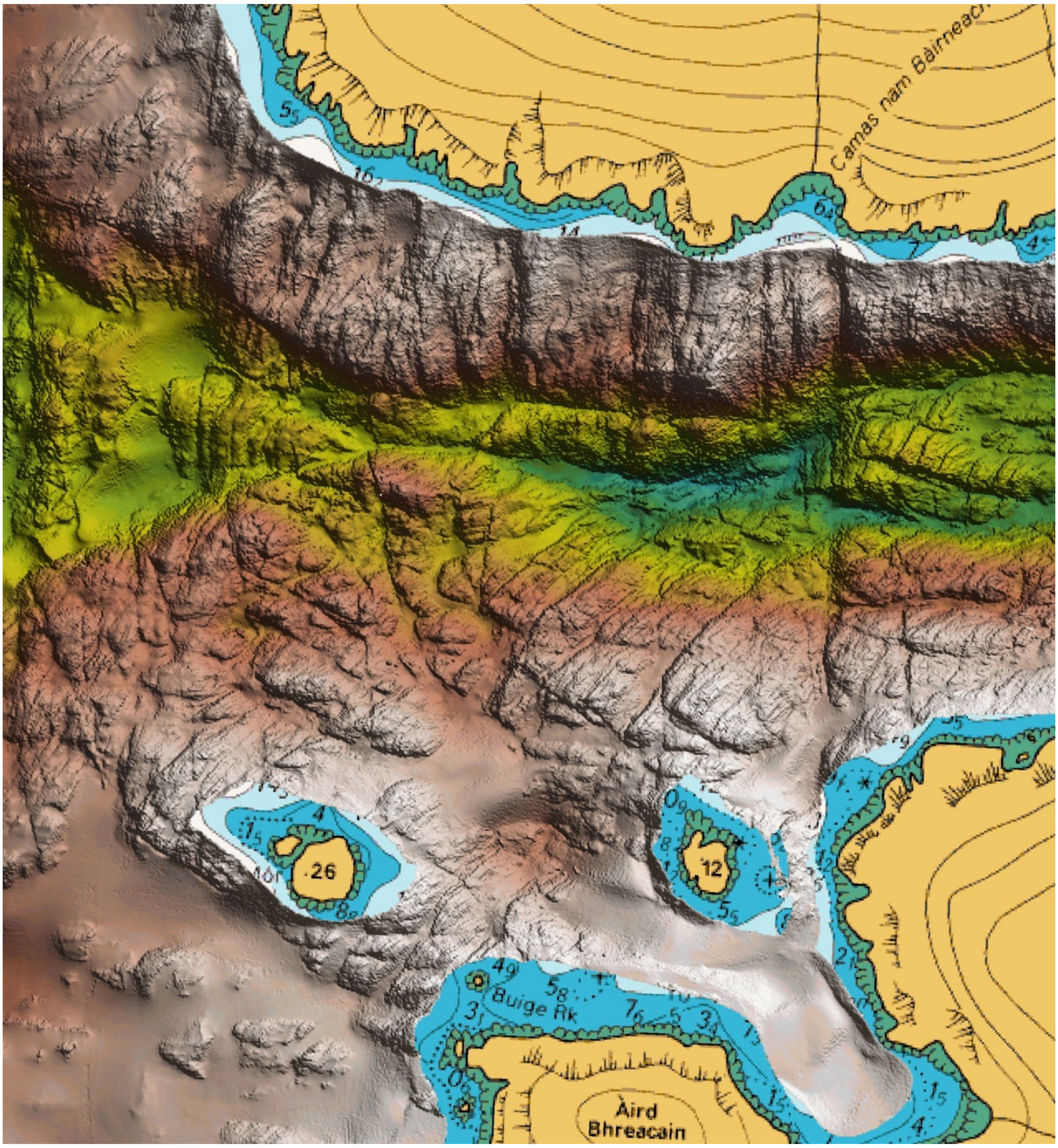


Figure 3.2. MAREMAP bathymetry chart from the Gulf of Corryvreckan. Note the impressive detail below 10m depth and complete absence of data from shallow areas.

3.2 Proposed field data collection methods

In this Section recommendations are presented for the design of surveys that return information to build the kelp GES indicator. SACFOR estimates of abundance of kelp at a number of depths at survey sites should be used at all times as the barest minimum level of data collection, to ensure continuity of data with the earlier MNCR datasets, and to enable comparison with the outputs from species and habitat models presented in Section 2. However, with a focus on the status of the populations of kelp species, it is possible to adopt more quantitative methods that would be more sensitive in detecting change over time. Categorical abundance methods require a large number of surveys to detect change with an acceptable degree of statistical power. A report for JNCC on the sensitivity of the method for detecting climate change effects on intertidal species suggested that at least 50 sites chosen at random along a coastline would be needed to detect an order of magnitude change, if these are not re-surveys of existing sites (Burrows *et al* 2006).

The suggested protocols include consideration of level of taxonomic resolution required: e.g. (i) only kelp to species, (ii) kelp to species plus conspicuous community members, (iii) full species ID for kelp and as many associated species as possible (using MNCR protocols and data recording sheets, and demanding highly trained personnel).

3.2.1 Sampling strategy

A two-tiered approach is outlined here and is comprised of (1) a series of sentinel sites surveyed regularly (ideally annually), and (2) broad-scale surveys to assess the state of kelp populations between the survey sites (more detail provided in Sections 3.2.2 – 3.2.4). In summary:

1) **Sentinel sites**: Diver surveys: initially annually
   3 locations in each of 6 regions, 2-3 sites per location = 30 to 50 sites
   Using transects, quadrats (recorded in situ, and/or using photography)

2) **Broadscale surveys** (gap filling):
   By SACFOR: every 5 to 10 years – a further 40 to 60 sites.
   By diver survey (using MNCR levels of taxonomic resolution), by ROV, drop down video

3.2.2 Sentinel site surveys

Kelp dominated assemblages are highly variable in space and time, being influenced by fluctuating factors such as storm events (Dayton *et al* 1992), oceanographic processes driving variability in temperature and nutrients (Dayton *et al* 1999; Wernberg *et al* 2013), and human-medicated processes acting across the land-sea interface (Connell *et al* 2008). Therefore, improving current understanding of patterns and drivers of spatio-temporal variability will be critical for determining relevant ‘baselines’ against which to detect change. While semi-quantitative measures (e.g. the SACFOR scale) are useful for characterising coarse habitat structure at broad spatial scales, fine-resolution quantitative data are required to detect ecologically-meaningful change in the structure of kelp forests (Irving & Northern 2012). As such, it is recommended that multiple sentinel monitoring sites are established within biogeographic regions, to collect robust quantitative data on the abundance of kelp species and other conspicuous members of the associated community. Monitoring activities should be cost-effective and not reliant on expert taxonomists, but powerful enough to examine temporal variability and detect change outside bounds of ‘natural variability’.

Survey design and site selection

We envisage the UK split into five or six regions for the purposes of these surveys:

1) Scotland, south-west, north-west, north-east;
2) Wales;
3) Northern Ireland;
4) South-west England and English Channel;
5) East England;
6) North-west England and Isle of Man.

Surveys would be based around three hub locations per region, with three to four sites surveyed per location.

Site survey methods – kelp community assessment by divers

Each sentinel monitoring site should be representative of local biogeographic conditions. As previously stated, cost-effective monitoring methods that enable the collection of robust data during just one diving day should be employed at these sites. This is assuming that the dive team consist of four appropriately-qualified divers (i.e. HSE standard or equivalent) who are capable of carrying out four separate paired-dives between them. Methods that quickly sample the different taxa of the entire community assemblage must be utilised so that the overall ecological structure of the kelp-dominated community can be determined. The following approach is recommended:

1) Two depths should be selected to set-up monitoring stations at each sentinel site; one to represent kelp forest (>20% cover) and one kelp park (<5% cover). As a result of variable light attenuation across the biogeographic regions and at specific sites the position of these station depths will vary between sentinel sites. Therefore, the depth of these stations will likely vary between 5–10m for the kelp forest and 10–15m for the kelp park.

2) If possible permanent underwater structures should be established at each station so that all data are collected from fixed location over the duration of the monitoring programme. This is initially costly (around £3,500 for construction and deployment of one station), however, there is general agreement across the scientific community that monitoring from fixed locations provides data that are considerably more sensitive to ecological change that those collected haphazardly (Lundälv *et al* 1986; Kingsford & Battershill 1998; Hill & Wilkinson 2004). When funds are limited then data may also be collected alternatively from haphazardly-positioned transects directly under the same position each year, although the sensitivity to change may not be as high as those collected from the fixed station.

3) Given the limits on what is achievable during one diving day, if sampling from two permanent stations at each sentinel site, one located between 10-15m depth and one 5–10m depth, it is likely that 4 – 5 transects (10m length) can be run out from the permanent markers at each station and sampled.

4) Each 10m transect will have the following data collected: diver one will have a 50x50cm open-ended quadrat to record the density and size of kelps (taxa; “juvenile” = 10-50cm; “adult” = >50cm) and abundance of mobile macro-invertebrates (i.e. urchins and crustacea) and diver two will have a 25x25cm photoquadrat to collect sample images of the sessile macro-invertebrate and macroalgal taxa. This will generate empirical data on kelp-dominated community structure and can be used to examine the relative abundances of key species (i.e. *Laminaria hyperborea* versus *Laminaria ochroleuca*). Similar approaches have been used to quantify kelp forest structure at multiple sites in south-west England (Figure 3.3).

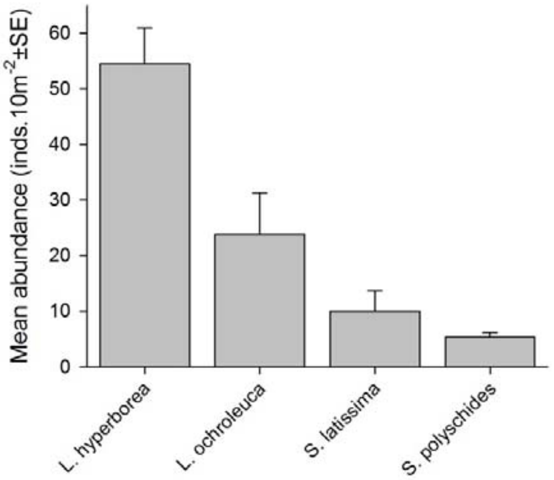


Figure 3.3. The abundances of kelp species at West Mewstone, south-west England, as determined by diver transects. Smale, unpublished data.

It is recommended that diving surveys form the basis of ongoing monitoring of indicators of GES. In addition to quantifying maximum kelp and kelp park depth, monitoring should involve the quantification of kelp forest extent and structure.

The survey activities would facilitate ground-truthing of the predictive modelling outlined in Section 2 and generate robust quantitative data on indicators of GES.

These recommendations are in line with other similar ones for surveying kelp habitats. Maggs and Bunker (unpublished report cited in Burton *et al* 2008), for example, recommended that:

1) “Monitoring should be carried out on bedrock; this meets the need for SAC monitoring of "reef". Algal data collected from pebble communities shows too much variation (dispersion in MDS plots) and therefore has no power to detect change over time. Data from algae on cobbles identified in lab did not exhibit this problem, but the methods are time-consuming and require a high level of taxonomic expertise. Collection of algal data from cobble communities should be discontinued.

2) The sub-littoral fringe should be omitted from monitoring as results highly variable, and there are probably insufficient species to acquire good discriminatory power. In addition, working conditions and accuracy in the sub-littoral fringe are highly susceptible to disruption by weather.

3) At each site, two or three depths should be selected. Some of these are marked already by permanent markers. Suitable depths would be 5m (upper infralittoral), 8m (lower infralittoral) and 11m (upper circalittoral). The two lower depths are probably more important.”

Site survey methods – kelp depth assessment by drop-down cameras

An alternative method to examine depth distributions and maximum depth of kelp park and kelp individuals is to use remotely deployed drop-cameras from a survey vessel (i.e. “half-decker” survey vessel with a dry cabin, power supply and reliable chart plotter and depth sounder). These surveys would run in tandem with the dive surveys at each sentinel site and should ideally be conducted on the same day. However, care should be taken to ensure that the drop-camera apparatus and the divers are sufficiently far apart as to not affect one another.

Imagery from the cameras can be used to assess percentage kelp canopy cover with relative ease, thus enabling an assessment of which areas contain kelp forest and kelp park. As previously mentioned, they can also operate over meso-scale areas, from 10–1000m^2, which suits the variable bathymetric profiles likely encountered at each sentinel site. At sites with a shallow profile, where the kelp forest and kelp park may be separated by 100's of metres, this offers a distinct advantage over diver-based methods, which are distance limited. The following approach is recommended for drop-survey methods:

1) At least 3 - 5 transects should be run along the top of the kelp canopy at each sentinel site, from the shore outwards along a course that is as perpendicular to the shore as possible, until the limit of kelp growth is established. Alternatively a bearing must be selected and followed for each transect.

2) All transects should be begin at the 5m depth contour inshore, or nearer if the skipper of the survey vessel is comfortable to go in closer to shore. At this point the drop camera is carefully lowered over the side of the vessel until it is hovering approximately 2m above the kelp canopy beneath, after which recording can begin. This height may change depending on the turbidity. It is more important to ensure that the field of view is adequate enough to record the following: % cover of kelp canopy, bare rock, unstable substratum, habitat forming/altering macroinvertebrate taxa.

3) Constant communication is required between the boat skipper, the camera operator viewing the imagery in the cabin, and the deck operator, who is lowering and raising the camera frame to the appropriate height as the boat begins to navigate the transect line as slowly as possible (1 knot or less if possible). However, local tidal drift and windage may not always allow for this.

4) As the boat progresses along the transect line, it is likely that too much will be going on to record ecological data. It is better to focus on recording accurate boat positions and depth (possible through the instant Man Over Board [MOB] chart plotter function), and noting when these occurred relative to real and camera time. A good team will quickly develop a good recording system with little time and practice. After the survey it is these data that are used to determine where all the relevant benthic features are and at what depth they occurred at.

3.2.3 Rapid broadscale surveys using SACFOR

Sentinel sites offer much in terms of added understanding of temporal variability, but this comes at a cost of lack of information about the fate of associated species, and importantly the fate of kelp populations at locations between the chosen sentinel sites. The MNCR approach (Hiscock 1996), with its carefully thought out system of survey and recording, and compatibility with the current database and entry system, offers a ready-made solution to this aspect of UK kelp monitoring.

Depending on the resources available and the question at hand (i.e. whether there is a local, regional or national focus to the assessment of GES), the broadscale survey should aim to cover as much of the coastline in the target area. If the approach of using the species distribution model residuals to indicate GES (Section 2.4.1) is adopted, then these surveys may be effectively clustered into 20km areas that, if possible, span a range of environmental conditions. Results from such surveys could then be assessed relative to model predictions by considering the proportion of sites within an area where abundance exceeds a threshold value.

3.2.4 Data collection methods

From the wide variety of monitoring methods available to kelp researchers the following methods, presented in this Section, have been selected as they are considered cost-effective, feasible and robust for the GES indicator tools. For a more detailed overview of the range of methods available for temperate sublittoral benthic monitoring see Kingsford and Battershill (1998) and for their spatial scale of operation see Van Rein *et al* (2009).

Diver survey using in-situ data collection

Divers have been used to survey the benthos in the UK since the 1970's and, in many ways, provide the most useful and cost-effective way to do so even today. Despite the health and safety concerns, training requirements, equipment costs and depth limitations, it is very difficult to achieve the same versatility and impression of the underwater world gained by a human diver, when used as a sampling platform. Therefore, the authors recommend that dive surveys form the basis of ongoing monitoring of indicators of GES. In addition to quantifying maximum kelp and kelp park depth, in situ diver monitoring should involve the quantification of kelp forest extent and structure. Diver surveys would also facilitate ground-truthing of the predictive modelling outlined in Section 2 and generate robust quantitative data on indicators of GES.

It is vital that consistency with earlier work is maintained to ensure that changes can be assessed relative to the earlier baseline. All collection of SACFOR data should follow the well documented methods of the Marine Nature Conservation Review (Connor and Hiscock, 1996).

Diver survey with photoquadrats

Where the size and density of attached macroalgae and benthic invertebrates permit, photoquadrats enable a rapid and efficient means of sample collection from marine benthic habitats (Bohnsack 1979; Preskitt *et al* 2004; Leujak & Ormond 2007). Depending on the methods of data extraction employed by the observer, highly objective quantification of epibenthic communities is possible (Van Rein *et al* 2011b). This may be achieved by extracting data from only the areas immediately under point-intercepts scattered across the sample image (Figure 3.4B). This removes any observer bias regarding abundance or percentage cover estimation and has been shown to improve the precision of data collected (see Drummond & Connell 2005, for discussion).

In the context of kelp-dominated habitats, photoquadrats may yet play a useful role as a rapid method for collecting large numbers of replicates within a short space of time (up to 53 replicates collected with 25x25cm photoquadrat during one dive to 10m and 40 replicates diving to 20m; Van Rein, unpublished). However, as the complex 3-dimensional structure of the kelp forest or park does not lend itself to be photographed only the sessile epifauna living in between the kelps (i.e. understorey community) could be physically sampled using photoquadrats. It is recommended that any kelp understorey monitoring using photoquadrats be conducted in tandem with kelp-specific monitoring so that the understorey community may be directly associated with the adjacent kelp canopy community.

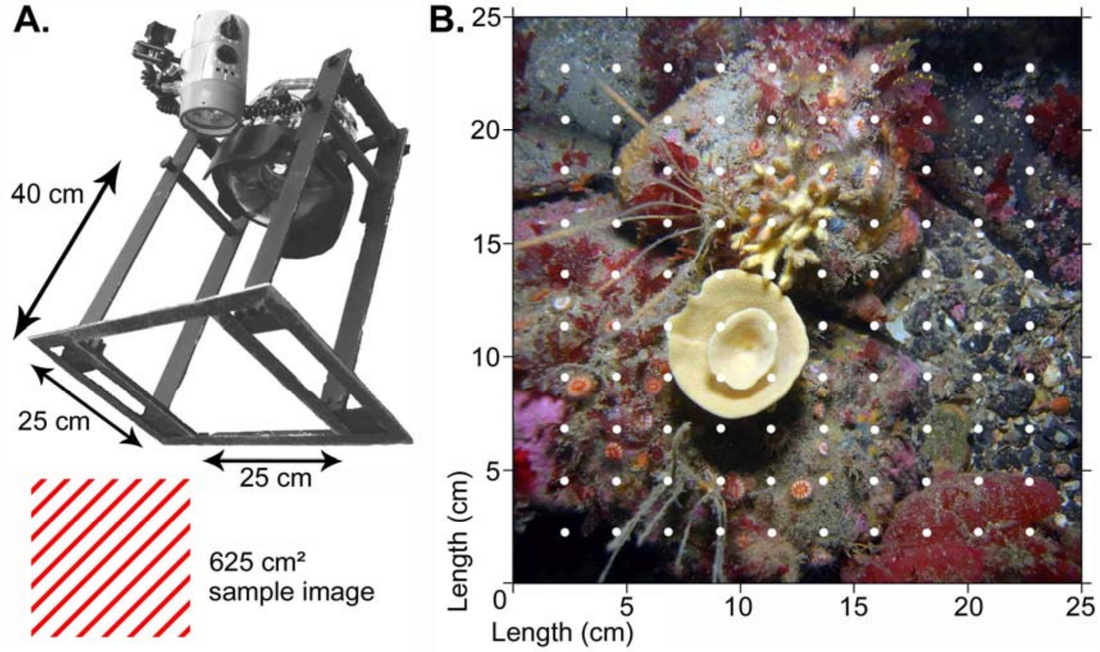


Figure 3.4. (A) Purpose-built photoquadrat used to collect benthic samples. The aluminium frame is constructed around a Nikon digital single-lens reflex (DSLR) camera in an Ikelite underwater housing with a single Ikelite strobe. (B) An example photoquadrat sample image collected from 30 m depth at Damicornis Bay, Rathlin Island in 2010. The 100 data extraction points are arranged in a regular 10 x 10 grid (indicated by white dots). Figure modified and reproduced from Van Rein *et al* 2011b.

Drop-down video camera and remotely operated vehicle (ROV) surveys

Drop cameras offer an excellent means of surveying marine habitats over the meso-scale (10–1000 m^2). If using a video-based approach, rather than stills, the data resolution suits the scale of operation in that only coarse taxa identification is possible (Van Rein *et al* 2009). This may reduce post-processing of video data. ROVs are generally more expensive to buy and may only result in data of comparable quality.

Drop camera drifts over areas of kelp can be easily repeated over time as long as geographical positions are accurately fixed and boat depth soundings taken regularly during the survey. Typically, the apparatus consists of a video camera and light capable of simultaneously recording digital imagery and displaying a live feed to the surface (e.g. Rovtech Systems Ltd), housed in a bespoke protective stainless steel frame which is then lowered from the survey vessel (Figure 3.5).

The video imagery collected during these surveys can be analysed in a number of cost-effective ways, such as rapid visual counts and fast visual counts (Michalopoulos *et al* 1992; Mitchell & Coggan 2007; Service & Goldring, 2007; Van Rein *et al* 2011a). Alternatively, a new method specific to kelp canopy cover could be developed through a pilot study. Once trialled this method should initially be the first level of survey at the sentinel sites and used to inform where the monitoring stations are to be located.

Figure 3.5. Drop camera, light and stainless steel frame ready for deployment to 20m depth off Rathlin Island, Northern Ireland.

3.2.5 Estimated survey costs for recommended monitoring methods

It is difficult to estimate survey costs without knowing what boat and diver costs are in each biogeographic region. Poor weather and equipment failure can increase overall costs while in-house training and survey teams can reduce it. Some costs are considered ‘one offs’, such as the construction and deployment of permanent station markers (complete with ultra short baseline trackers and topside units), while maintenance and staff costs are steady. Bearing all of this in mind we have provided a rough estimate of some potential costs that could be incurred in a year when 10 broadscale sites and 54 sentinel sites are surveyed across the UK (Table 3.1). It is worth noting that this is likely the minimum spend necessary to achieve accurate assessments of GES in kelp habitats throughout the UK. It is also worth noting that not all sites listed in Table 3.1 are located in the one biogeographic region, but are located across the whole UK.

Table 3.1. Estimated survey and data extraction costs of recommended monitoring methods used to collect data for indicators of Good Environmental Status in kelp-dominated benthic habitats.

Monitoring sites	Method	Survey cost (per day)	Sampling effort (days.year-1)	Total survey cost	Data extraction cost (per day)	Extraction effort (days.yr-1)	Total extraction cost	Combined cost
Sentinel sites	Diver surveys	£1,500	54	£81,000	£200.00	54	£10,800	£91,800
	Drop camera	£1,000	54	£54,000	£200.00	54	£10,800	£64,800
Broadscale sites	Diver surveys	£1,500	10	£15,000	£200.00	10	£2,000	£17,000
	Drop camera	£1,000	10	£10,000	£200.00	10	£2,000	£12,000
Total		**£5,000**	**128**	**£160,000.00**	**£800**	**128**	**£25,600.00**	**£185,600**

3.2.6 Other survey techniques

Alongside the direct recording of kelp biota by divers, either quadrat-based numerical counts of stipe density or SACFOR-based assessments of multiple, further methodologies are possible. The use of these in place of the ones suggested above warrants further research and side-by-side comparison with more established surveys methods, to maximise the comparability of data obtained.

For example, if time/cost constraints allow, it would be highly beneficial to quantify and monitor kelp biomass over time, to detect ecologically relevant shifts in population structure (driven by storms, for example). Here, multiple haphazardly placed ($n \geq 5$) 1m^2 quadrats should be placed at the permanent stations of each sentinel site and all kelp removed and collected for analysis. Kelp identification and fresh weight could be completed immediately after the dive (using spring scales) and the material returned.

3.2.7 A comparison among survey methods is needed

It is clear that there are a growing number of possible methods for surveying kelp populations and associated sublittoral benthos on hard substrata. The proponents of each method are often quick to dismiss the benefits of other approaches. A straightforward cross-calibration of different techniques, and an assessment of the efficacy of each, is needed for stronger recommendations to be made. This comparison should address questions of (i) accuracy, (ii) repeatability (among different teams using the same approaches), and (iii) representativeness.

Formal statistical power analysis offers a way to address issues of the usefulness of techniques in a monitoring programme (Burrows *et al* 2006), since it shifts the emphasis on to defining the amount of change or difference that the programme should detect. It is highly likely that different techniques will emerge as better suited to answering specific questions. Such analyses would bring much needed objectivity to the debates over the most appropriate approach.

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Appendix 1 – Abundance Categories

Abundance categories for species in the UK Marine Nature Conservation Review (Hiscock, 1996): S - Super-Abundant; A - Abundant; C - Common; F - Frequent; O -Occasional; R - Rare. Categories depended on percentage cover for space occupying species, and changed according to growth form, with a separate classification for those species recorded by population density.

	Growth form		Size of individuals / colonies					
% Cover	Crust/ Meadow	Massive /Turf	<1cm	1-3 cm	3-15 cm	>15 cm	Density	
>80%	S	S					>1/0.0001m^2 (1x1cm)	>10,000/m^2
40-79%	A	S	A	S			1-9/0.001m^2 (3.16x3.16cm)	1000-9999/m^2
20-39%	C	A	C	A	S		1-9/0.01m^2 (10x10cm)	100-999/m^2
10-19%	F	C	F	C	A	S	1-9/0.1m^2	10-99/m^2
5-9%	O	F	O	F	C	A	1-9/m^2	
1-5% or density	R	O	R	O	F	C	1-9/10m^2 (3.16x3.16m)	
<1% or density		R		R	O	F	1-9/100m^2 (10x10m)	
					R	O	1-9/1000m^2 (31.6x31.6m)	
						R	>1/10,000m^2 (100x100m)	<1/1000m^2

Appendix 2 – Statistical Methods

The details of methods of statistical model selection and summaries of models fitted to MNCR kelp data are presented here. All models were fitted using the `polr()` function in the R statistics package (version 2.15.0). The `polr()` function fits a logistic regression model to an ordered factor response, in this case the ordered abundance categories on the SACFORN scale. POLR stands for proportional odds logistic regression.

A range of alternatively formulated models were fitted. These models included effects of depth (`avgd`), SST in either February (`FebSSTvals`) or August (`AugSSTvals`), estimated chlorophyll *a* concentration (as log10 mg/m3, `lchla`), and wave fetch (log10 km) as a linear (`wf2`) and a quadratic term (`wf2q`, as the `wf2` value squared). The models had different number of interaction terms (where the response to one variable is modified by the value of a second variable): none (`olrk1`), one (`olrk6, 6a, 7 and 8`), two (`olrk3, 4` and `5`) and three (`olrk2`). Interaction between variables is denoted by a colon separating the two variable names.

Fit of different models was compared using AIC values (Akaike's Information Criterion), a combined index of the goodness of fit of the model and the number of parameters. Lower values of AIC indicate better fit of the model to the data after penalising for the positive effect of the number of parameters on the goodness of fit.

Models

```
olrk1<-polr(data=olrdatanm, nKelp ~ avgd + FebSSTvals + lchla + wf2, Hess
       = TRUE)
olrk2<-polr(data=olrdatanm, nKelp ~ avgd + FebSSTvals + lchla + wf2 +
       avgd:lchla + avgd:wf2 + lchla:wf2, Hess = TRUE)
olrk3<-polr(data=olrdatanm, nKelp ~ avgd + FebSSTvals + lchla + wf2 +
       avgd:lchla + lchla:wf2, Hess = TRUE)
olrk4<-polr(data=olrdatanm, nKelp ~ avgd + FebSSTvals + lchla + wf2 +
       avgd:lchla + avgd:wf2, Hess = TRUE)
olrk5<-polr(data=olrdatanm, nKelp ~ avgd + FebSSTvals + lchla + wf2 +
       avgd:wf2 + lchla:wf2, Hess = TRUE)
olrk6<-polr(data=olrdatanm, nKelp ~ avgd + FebSSTvals + lchla + wf2 +
       avgd:lchla , Hess = TRUE)
olrk6a<-polr(data=olrdatanm, nKelp ~ avgd + FebSSTvals + lchla + wf2 +
       wf2q + avgd:lchla , Hess = TRUE)
olrk7<-polr(data=olrdatanm, nKelp ~ avgd + FebSSTvals + lchla + wf2 +
       avgd:wf2, Hess = TRUE)
olrk8<-polr(data=olrdatanm, nKelp ~ avgd + FebSSTvals + lchla + wf2 +
       lchla:wf2, Hess = TRUE)

olrk11<-polr(data=olrdatanm, nKelp ~ avgd + AugSSTvals + lchla + wf2, Hess
       = TRUE)
olrk12<-polr(data=olrdatanm, nKelp ~ avgd + AugSSTvals + lchla + wf2 +
       avgd:lchla + avgd:wf2 + lchla:wf2, Hess = TRUE)
olrk13<-polr(data=olrdatanm, nKelp ~ avgd + AugSSTvals + lchla + wf2 +
       avgd:lchla + lchla:wf2, Hess = TRUE)
olrk14<-polr(data=olrdatanm, nKelp ~ avgd + AugSSTvals + lchla + wf2 +
       avgd:lchla + avgd:wf2, Hess = TRUE)
olrk15<-polr(data=olrdatanm, nKelp ~ avgd + AugSSTvals + lchla + wf2 +
       avgd:wf2 + lchla:wf2, Hess = TRUE)
olrk16<-polr(data=olrdatanm, nKelp ~ avgd + AugSSTvals + lchla + wf2 +
       avgd:lchla , Hess = TRUE)
olrk16a<-polr(data=olrdatanm, nKelp ~ avgd + AugSSTvals + lchla + wf2 +
       wf2q + avgd:lchla , Hess = TRUE)
olrk17<-polr(data=olrdatanm, nKelp ~ avgd + AugSSTvals + lchla + wf2 +
       avgd:wf2, Hess = TRUE)
olrk18<-polr(data=olrdatanm, nKelp ~ avgd + AugSSTvals + lchla + wf2 +
       lchla:wf2, Hess = TRUE)
```

AIC values for *Laminaria hyperborea* models

Models `olrk1` to `olrk8` included February SST as a predictor, while `olrk11` to `olrk18` used August SST. AIC values show that a model that included August SST and a quadratic term for wave fetch gave the best prediction of *Laminaria hyperborea* abundance (`olrk16a`).

```
       df      AIC
olrk1  10 10944.82
olrk2  13 10913.06
olrk3  12 10914.55
olrk4  12 10913.03
olrk5  12 10944.87
olrk6  11 10915.41
olrk6a 12 10889.32
olrk7  11 10943.11
olrk8  11 10946.47
```

```
        df      AIC
olrk11  10 10733.92
olrk12  13 10710.00
olrk13  12 10710.06
olrk14  12 10708.01
olrk15  12 10734.87
olrk16  11 10708.08
olrk16a 12 10699.94
olrk17  11 10733.98
olrk18  11 10735.01
```

```
> summary(olrk16a)
Call:
polr(formula = nKelp ~ avgd + AugSSTvals + lchla + wf2 + wf2q +
    avgd:lchla, data = olrdatanm, Hess = TRUE)

Coefficients:
             Value Std. Error t value
avgd        0.1213    0.00519  23.363
AugSSTvals -0.3140    0.02267 -13.852
lchla      -1.0898    0.28980  -3.761
wf2         2.7917    0.51110   5.462
wf2q       -0.2518    0.07957  -3.165
avgd:lchla  0.1538    0.02992   5.140

Intercepts:
  Value    Std. Error t value
0|1   1.2351   0.8717     1.4168
1|2   1.4021   0.8717     1.6084
2|3   1.7315   0.8717     1.9863
3|4   2.0349   0.8717     2.3343
4|5   2.5147   0.8719     2.8841
5|6   4.5239   0.8756     5.1666

Residual Deviance: 10675.94
AIC: 10699.94
```

AIC values for *Saccharina latissima* models

AIC values show that models that included August SST gave better prediction of *Saccharina latissima* abundance than those including February SST. The best of these models (`olrk14`) included interactions of depth and chlorophyll *a;* and depth and wave fetch. However, the model that included only the interaction of chlorophyll *a* and depth, and a quadratic term for wave fetch (`olrk16a`), was selected for ease of comparison with the same model for *Laminaria hyperborea*. Complex interactions make interpretation difficult so we opted not to fit the marginally better model in this case.

```
       df      AIC
olrk1  10 10561.51
olrk2  13 10470.77
olrk3  12 10536.66
olrk4  12 10471.91
olrk5  12 10493.87
olrk6  11 10535.12
olrk6a 12 10423.07
olrk7  11 10492.84
olrk8  11 10563.26
```

```
        df      AIC
olrk11  10 10375.19
olrk12  13 10276.71
olrk13  12 10355.48
olrk14  12 10275.72
olrk15  12 10295.19
olrk16  11 10353.57
olrk16a 12 10283.40
olrk17  11 10293.21
olrk18  11 10376.91
```

```
> summary(olrk16a)
Call:
polr(formula = nKelp ~ avgd + AugSSTvals + lchla + wf2 + wf2q + 
    avgd:lchla, data = olrdatanm, Hess = TRUE)

Coefficients:
             Value Std. Error  t value
avgd        0.1274   0.005768  22.0903
AugSSTvals -0.2797   0.023903 -11.7024
lchla       0.1537   0.285598   0.5383
wf2         3.5018   0.468283   7.4780
wf2q       -0.6410   0.076145  -8.4176
avgd:lchla  0.1497   0.032567   4.5972

Intercepts:
    Value    Std. Error t value
0|1  -0.2430   0.8004    -0.3036
1|2   0.0021   0.8005     0.0026
2|3   0.6509   0.8008     0.8128
3|4   1.3120   0.8010     1.6380
4|5   2.0627   0.8017     2.5729
5|6   4.3452   0.8198     5.3003

Residual Deviance: 10259.40 
AIC: 10283.40
```

AIC values for *Alaria esculenta* models

AIC values showed that models that included August SST gave better prediction of *Alaria esculenta* abundance than those including February SST. For this species, the quadratic term for wave exposure added no predictive ability to the models, so the corresponding model to those fitted to *Laminaria hyperborea* and *Saccharina latissima* (`olrk16`).

```
       df      AIC
olrk1  10 2308.088
olrk2  13 2296.050
olrk3  12 2298.362
olrk4  12 2294.164
olrk5  12 2308.160
olrk6  11 2296.368
olrk6a 12 2296.837
olrk7  11 2306.450
olrk8  11 2309.849
```

```
        df      AIC
olrk11  10 2186.031
olrk12  13 2176.138
olrk13  12 2178.037
olrk14  12 2175.598
olrk15  12 2184.797
olrk16  11 2176.651
olrk16a 12 2178.470
olrk17  11 2184.736
olrk18  11 2186.430
```

```
> summary(olrk16)

Call:
polr(formula = nKelp ~ avgd + AugSSTvals + lchla + wf2 + avgd:lchla, 
    data = olrdatanm, Hess = TRUE)

Coefficients:
             Value Std. Error t value
avgd        0.1867    0.01491 12.5158
AugSSTvals -0.5533    0.05567 -9.9376
lchla      -0.5172    0.62494 -0.8275
wf2         1.8519    0.14793 12.5191
avgd:lchla  0.3318    0.10195  3.2541

Intercepts:
  Value   Std. Error t value
0|1 -0.1784  0.8711    -0.2048
1|2  0.1054  0.8713     0.1209
2|3  0.6146  0.8728     0.7042
3|4  1.0455  0.8751     1.1947
4|5  1.5836  0.8797     1.8001
5|6  3.4816  0.9441     3.6879

Residual Deviance: 2154.651 
AIC: 2176.651
```

AIC values for *Laminaria digitata* models

Model fit indices for *Laminaria digitata* showed that February SST was a slightly better predictor of abundance than August SST. The best model was one that did not include a quadratic term for wave fetch, but did have an interaction term for chlorophyll *a* and depth (`olrk6`).

```
       df      AIC
olrk1  10 1942.917
olrk2  13 1944.709
olrk3  12 1943.094
olrk4  12 1942.888
olrk5  12 1945.849
olrk6  11 1941.341
olrk6a 12 1943.262
olrk7  11 1944.397
olrk8  11 1944.347
```

```
        df      AIC
olrk11  10 1952.272
olrk12  13 1953.655
olrk13  12 1952.265
olrk14  12 1952.027
olrk15  12 1954.622
olrk16  11 1950.765
olrk16a 12 1952.440
olrk17  11 1953.508
olrk18  11 1953.344
```

```
Call:
polr(formula = nKelp ~ avgd + FebSSTvals + lchla + wf2 + avgd:lchla, 
    data = olrdatanm, Hess = TRUE)

Coefficients:
      Value Std. Error t value
avgd       0.3151    0.02681  11.753
FebSSTvals 0.3680    0.10835   3.396
lchla      1.2068    0.64357   1.875
wf2        0.2948    0.10210   2.887
avgd:lchla 0.2608    0.14111   1.849

Intercepts:
    Value   Std. Error t value
0|1  5.3079  0.9370     5.6646
1|2  5.4833  0.9378     5.8470
2|3  5.7740  0.9391     6.1481
3|4  6.2358  0.9419     6.6202
4|5  6.8389  0.9477     7.2162
5|6  8.5967  1.0010     8.5882

Residual Deviance: 1919.341 
AIC: 1941.341
```

AIC values for *Saccorhiza polyschides* models

The best model fitted to *Saccorhiza polyschides* data included February SST as a predictor and an interaction term for chlorophyll *a* and depth.

```
       df      AIC
olrk1  10 4035.986
olrk2  13 4017.502
olrk3  12 4016.555
olrk4  12 4038.479
olrk5  12 4016.751
olrk6  11 4037.888
olrk6a 12 3977.523
olrk7  11 4036.674
olrk8  11 4015.965

        df      AIC
olrk11  10 4073.877
olrk12  13 4066.164
olrk13  12 4064.287
olrk14  12 4077.585
olrk15  12 4064.743
olrk16  11 4075.829
olrk16a 12 4015.889
olrk17  11 4075.647
olrk18  11 4062.913
```

```
> summary(olrk6a) # Best AIC model for Sapol
Call:
polr(formula = nKelp ~ avgd + FebSSTvals + lchla + wf2 + wf2q + 
    avgd:lchla, data = olrdatanm, Hess = TRUE)

Coefficients:
       Value Std. Error t value
avgd        0.11654    0.00939 12.4103
FebSSTvals  0.48942    0.07714  6.3447
lchla      -0.99068    0.56845 -1.7428
wf2         7.11063    0.89629  7.9334
wf2q       -1.04414    0.13863 -7.5320
avgd:lchla  0.03337    0.05867  0.5687
```

```
Intercepts:
    Value   Std. Error t value
0|1 16.8391  1.5723    10.7097
1|2 17.0462  1.5730    10.8370
2|3 17.7168  1.5749    11.2492
3|4 18.3894  1.5772    11.6596
4|5 19.3345  1.5823    12.2196
5|6 21.2948  1.6268    13.0897

Residual Deviance: 3953.523
AIC: 3977.523
```