

Case Report for kelp forests habitat. OSPAR 787/2021

Thibaut de Bettignies, Claire Hébert, Jorge Assis, Inka Bartsch, Trine Bekkby, H. Christie, Karsten Dahl, Sandrine Derrien-Courtel, Hugh Edwards, Karen Filbee-Dexter, et al.

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OSPAR COMMISSION

Case report for kelp forest habitat



2021

Biodiversity and Ecosystems Series

Case report for kelp forest habitat

Citation

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OSPAR Convention

The Convention for the Protection of the Marine Environment of the North-East Atlantic (the "OSPAR Convention") was opened for signature at the Ministerial Meeting of the former Oslo and Paris Commissions in Paris on 22 September 1992. The Convention entered into force on 25 March 1998. The Contracting Parties are Belgium, Denmark, the European Union, Finland, France, Germany, Iceland, Ireland, Luxembourg, the Netherlands, Norway, Portugal, Spain, Sweden, Switzerland and the United Kingdom.

Convention OSPAR

La Convention pour la protection du milieu marin de l´Atlantique du Nord-Est, dite Convention OSPAR, a été ouverte à la signature à la réunion ministérielle des anciennes Commissions d´Oslo et de Paris, à Paris le 22 septembre 1992. La Convention est entrée en vigueur le 25 mars 1998. Les Parties contractantes sont l´Allemagne, la Belgique, le Danemark, l´Espagne, la Finlande, la France, l´Irlande, l´Islande, le Luxembourg, la Norvège, les Pays-Bas, le Portugal, le Royaume- Uni de Grande Bretagne et d´Irlande du Nord, la Suède, la Suisse et l´Union européenne

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a) Nomination

Kelp forest habitat dominated by the following species (per OSPAR Region):

Alaria esculenta (in OSPAR Region II, and the French EEZ portion of region III)

Laminaria digitata (in OSPAR Region II, IV and the French EEZ portion of region III)

Laminaria hyperborea (in OSPAR Region IV)

Laminaria ochroleuca (in OSPAR Region IV)

Saccharina latissima (in OSPAR Region II, IV)

Saccorhiza polyschides (in OSPAR Region IV)

These six species form kelp forest habitat for which there is firm evidence of decline in the relevant parts of the OSPAR region or the specific portion identified. Five other species form kelp foresthabitats in the OSPAR area but are not known to be in decline: *Agarum clathratum*, *Laminaria solidungula, Hedophyllum nigripes* (formerly *Saccharina nigripes* or *Laminaria nigripes*)*, Saccorhiza dermatodea* (see BDC 20/7/1 Annex 11)[1].

b) Reason for review

Drastic losses of kelp forest habitat have already occurred in the southern part of the OSPAR area with significant declines at several locations elsewhere. Considering the sensitivity of this complex and highly productive habitat to cumulative effects of multiple pressures, and the increasing threat posed by climate change, the nominated kelp forest habitat, as defined below, should be added to the OSPAR List of Threatened and/or Declining habitats.

c) Definition for habitat mapping

'Kelp forests' are defined as complex habitats created by large brown seaweeds mostly in the order Laminariales and some species of the order Tilopteridales (Bolton, 2016; Wernberg & Filbee-Dexter, 2019). They typically occur from the low intertidal down to shallow subtidal zones (approximately 40 m depth, depending on the water clarity with a depth limit at around 10% of incoming irradiance: Lüning, 1990) of temperate and polar coastal waters. As surface seawater temperature increases towards lower latitudes, kelp forests become excluded from shallow areas and restricted to deeper, cooler water (Graham *et al.*, 2007; Marzinelli *et al.*, 2015).

A kelp forest has a canopy formed by kelp stipes that hold fronds tens of centimetres to several metres above the substratum. This modifies the local environment to support a specific and distinct associated understorey and epibiotic community, which would not persist without the canopy (Fig. 1) (Flukes *et al.*, 2014; Leclerc *et al.*, 2015; Teagle *et al.*, 2017). A forest exhibits dense stands with the fronds forming an almost complete canopy (often referred as Leaf Area Index > 1; Pehlke & Bartsch, 2008) equivalent to having a stipe density of at least 3-5 adult stipes m^{-2} for the kelp species of the North-East Atlantic (Fig. 1). Depending on the latitude and environmental conditions, kelp forests have

[1] The species *Undaria pinnatifida* can also form kelp forest habitats but was not assessed against the Texel-Faial Criteria summarized in BDC 20/7/1 Annex 11 because of its invasive and non-native characteristics.

different kelp species assemblages and may show density variations for example after storms or winter die-back of annual sporophytes. Six canopy-forming species that may be locally dominant or occur in mixed assemblages with varying densities and are able to form dense canopies are considered here: the Laminariales *Alaria esculenta, Laminaria digitata, L. hyperborea, L. ochroleuca, Saccharina latissima* (formerly *Laminaria saccharina*), and the Tilopteridale *Saccorhiza polyschides* (Fig. 1).

An indicative list of corresponding EUNIS (2012) habitat codes is provided in Appendix I as a tool for mapping and classification exercise.

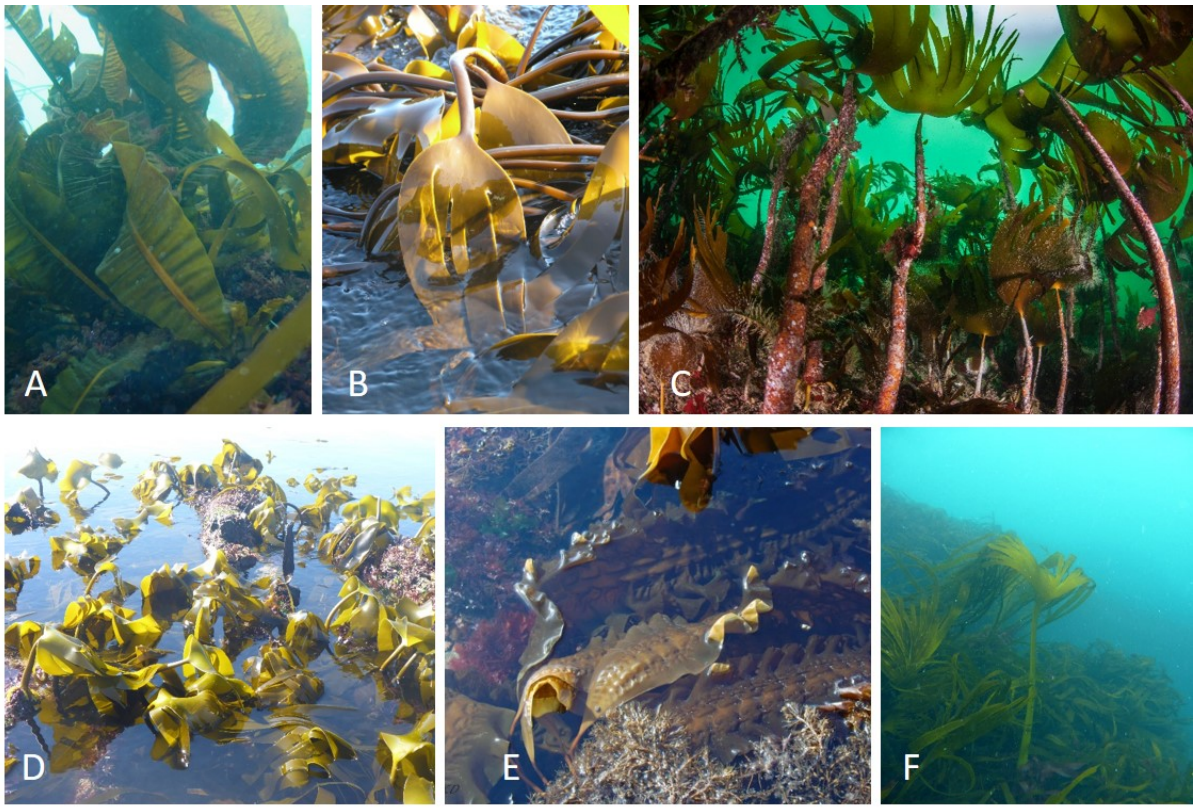


Figure 1. North-East Atlantic kelp forests dominated by the following species: A. *Alaria esculenta* (© MNHN Concarneau - R. Derrien), **B.** *Laminaria digitata* (© AWI Helmholtz Center for Polar and Marine Research - M. Molis), **C.** *L. hyperborea* (© SNH), **D.** *L. ochroleuca* (© CIIMAR - J. Franco), **E.** *Saccharina latissima* (© MNHN Concarneau - R. Derrien) and **F**. *Saccorhiza polyschides* (© MNHN Concarneau - A. Le Gal).

d) Geographic extent

a. OSPAR Regions

All OSPAR Regions

In the North-East Atlantic, kelp forests dominate shallow subtidal rocky reefs except in the most sheltered or turbid locations in northern-central Europe (Smale *et al.,* 2013; Araújo *et al.,* 2016) and become rare and more restricted to deeper waters along coastlines south of northwest Iberia (Assis *et al.*, 2017, 2018a). In the warm temperate Azores archipelago (OSPAR region V) or the shallow seamounts off southwestern Iberia (offshore Region IV), they occur locally at depths of more than 40 metres.

In the OSPAR area, the six listed kelp species form kelp forest habitats (as defined above) and are among the most dominant seaweeds along the European coastline. Found in the north from the Svalbard islands in Arctic Norway, the Murmansk coast of Russia, Greenland and Iceland through to Portugal and offshore to the Azores in the south, kelp forests occur in all five OSPAR Regions (Araújo *et al.*, 2016; Krumshansl *et al.,* 2016; Filbee-Dexter *et al.*, 2019) (Fig. 2). Their distribution and abundance vary with latitude, according to their thermal affinities (Lüning, 1990; Müller *et al.,* 2009; Araújo *et al.,* 2016, Assis *et al.,* 2017). Forests of *Laminaria hyperborea* and *Saccharina latissima* are the two most widely distributed kelp forests along the rocky coast of the North-East Atlantic Ocean: from Portugal to the Svalbard archipelago (Fig. 2). *Laminaria hyperborea* typically occurs in more wave-exposed conditions (Bekkby *et al.*, 2009) and *S. latissima* in medium/sheltered conditions and in brackish water (Bekkby & Moy, 2011). The Lusitanean species *Laminaria ochroleuca* presents a warmer affinity and is present from Ireland down to OSPAR lowest latitudes, namely on the Gorringe seamount (off southwestern Iberia) and the Azores archipelago

b. Biogeographic zones

Barents Sea (including Svalbard), Norwegian coast, North-East Greenland shelf, South East Greenland – North Iceland Shelf, South Iceland-Faeroe Shelf, Boreal, Boreal-Lusitanean, Lusitanean-Boreal, Warm/Cool Lusitanean, Macaronesian Azores.

e) Application of the Texel-Faial criteria

Texel-Faial criteria are applied to the nominated habitat as defined in the 'c) Definition for habitat mapping' section.

a. Global importance

Yes

Kelps are found in 43% of the world's marine ecoregions along coastlines of all continents except Antarctica (Krumhansl *et al.*, 2016), and kelp forests are the dominant biogenic habitat along around 25% of the world's coastline by length (Wernberg *et al.*, 2019). However, Antarctica is inhabited by big brown algae of the order Desmarestiales and thereby provide a similar habitat (e.g., *Himantothallus grandifolius* in King George Island, Valdivia *et al.,* 2014). The global importance of the OSPAR region for the kelp forest habitat as a whole is very high as some of these 'forest-forming species' distributions are limited to the North-East Atlantic (e.g., *Laminaria hyperborea, L. ochroleuca, and Saccorhiza polyschides*). Other species have a wide circumpolar distributions in the northern hemisphere (e.g., *Saccharina latissima*, *Alaria esculenta* and *Laminaria digitata*), but still a large part of the total global distribution is found within the OSPAR area (e.g., *Alaria esculenta*, *Laminaria*

digitata, *L. hyperborea* and *Saccharina latissima*) (GBIF, 2019; OBIS, 2019). In general, the dominant canopy formers in the North-East Atlantic belong to the genus *Laminaria* (e.g., *Laminaria hyperborea*, *L. digitata* and *L. ochroleuca*) (Bartsch *et al.*, 2008; Bolton, 2010; Smale *et al.*, 2013), with a high proportion of these kelp forests found within OSPAR waters (GBIF, 2019; OBIS, 2019). Because such kelp forests are dominated by one or very few species in the North-East Atlantic, their decline will thus lead to the loss of kelp forest habitat.

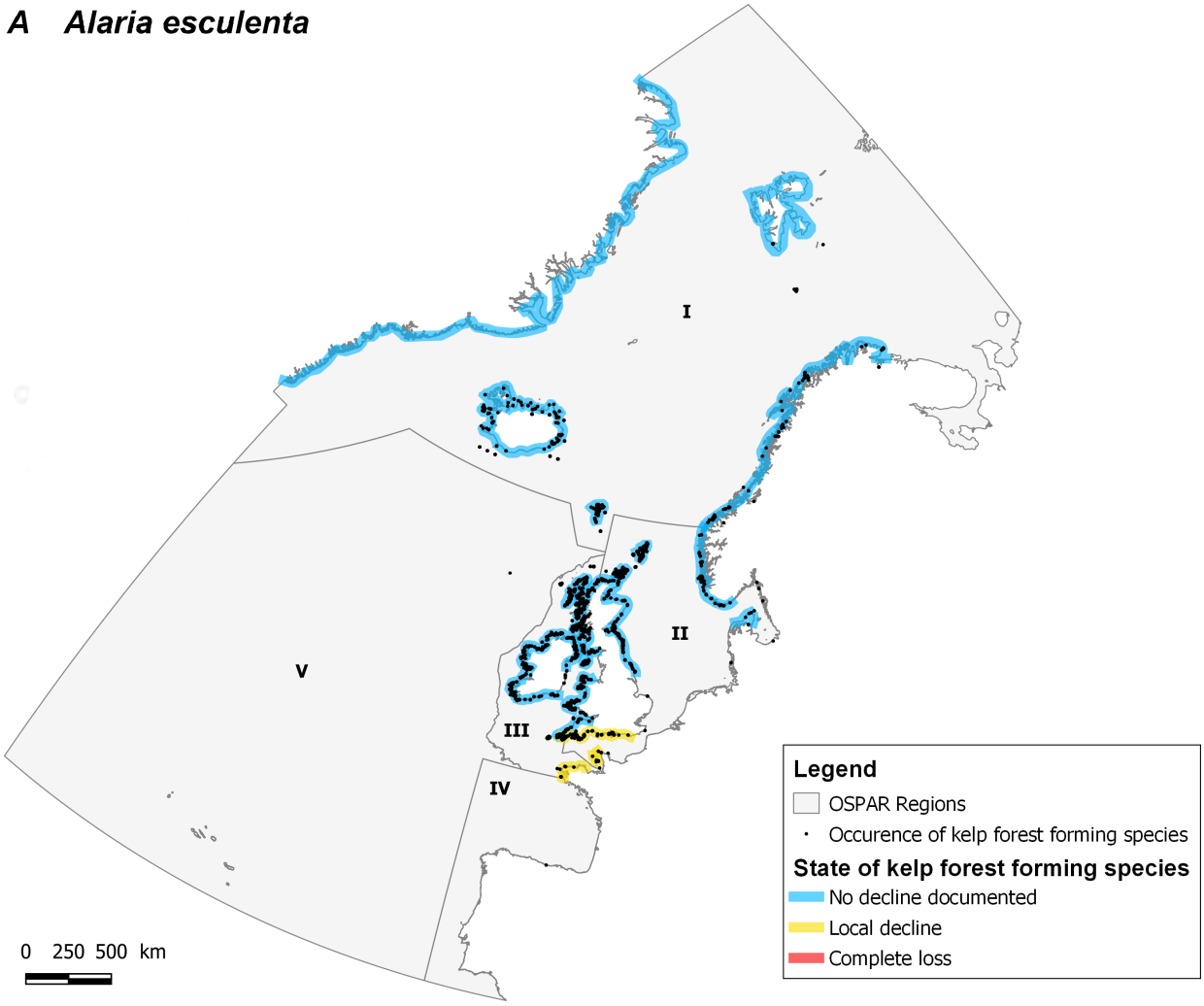
b. Regional importance

Yes

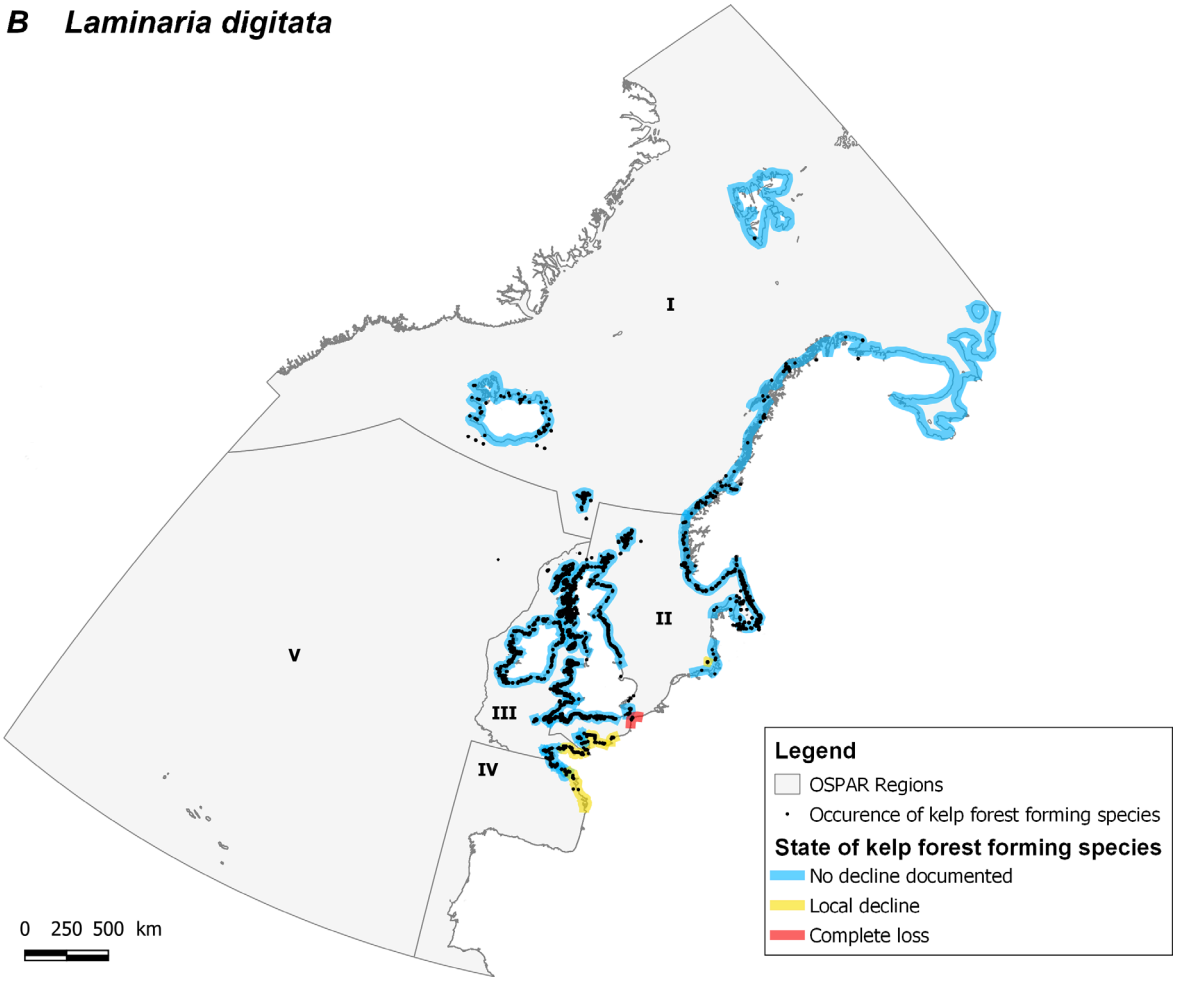
Throughout the OSPAR area, kelp forests vary in terms of composition, biomass and dominant species. While some forest-forming species are widespread throughout North-East Atlantic waters (e.g., *Laminaria hyperborea, L. digitata* and *Saccharina latissima*), others are present only in certain OSPAR regions. This is the case for *Laminaria ochroleuca* and *Saccorhiza polyschides*, which are respectively warm temperate and temperate water Lusitanian species primarily found in OSPAR Regions III and IV (Fig. 2 ; Appendix II). Recent research underlined the fundamental role of populations of those two species, located in the OSPAR area as they retain some of the largest genetic diversity relative to the remaining species distribution (Assis *et al.,* 2016a and 2018b). *Alaria esculenta* and *Laminaria digitata,* two Arctic cold-temperate amphioceanic kelp species, both occur mainly in Regions I, II and III (Fig. 2; Appendix II).

Figure 2: **Distribution maps of six nominated kelp forest habitat-forming species** (**A.** *Alaria esculenta*, **B.** *Laminaria digitata*, **C.** *L. hyperborea**, **D.** *L. ochroleuca*, **E.** *Saccharina latissima* and **F**. *Saccorhiza polyschides).* Colour lines display the status of kelp forests. Blue lines show kelp forest habitat ranges with no documented evidence of decline, yellow lines show regions of local decline and red lines show complete loss of kelp forest habitat. Occurrence data (black dots) are from the Global Biodiversity Information Facility (GBIF, 2019), the Ocean Biogeographic Information System (OBIS, 2019) and data provided by Contracting Parties (UK, Sweden, Norway and France). **NB L. hyperborea kelp forest habitats are marked yellow in region I, but are not part of the nomination as this habitat recovered in northern Norway over the last 30 years due to reduced sea urchin grazing.*

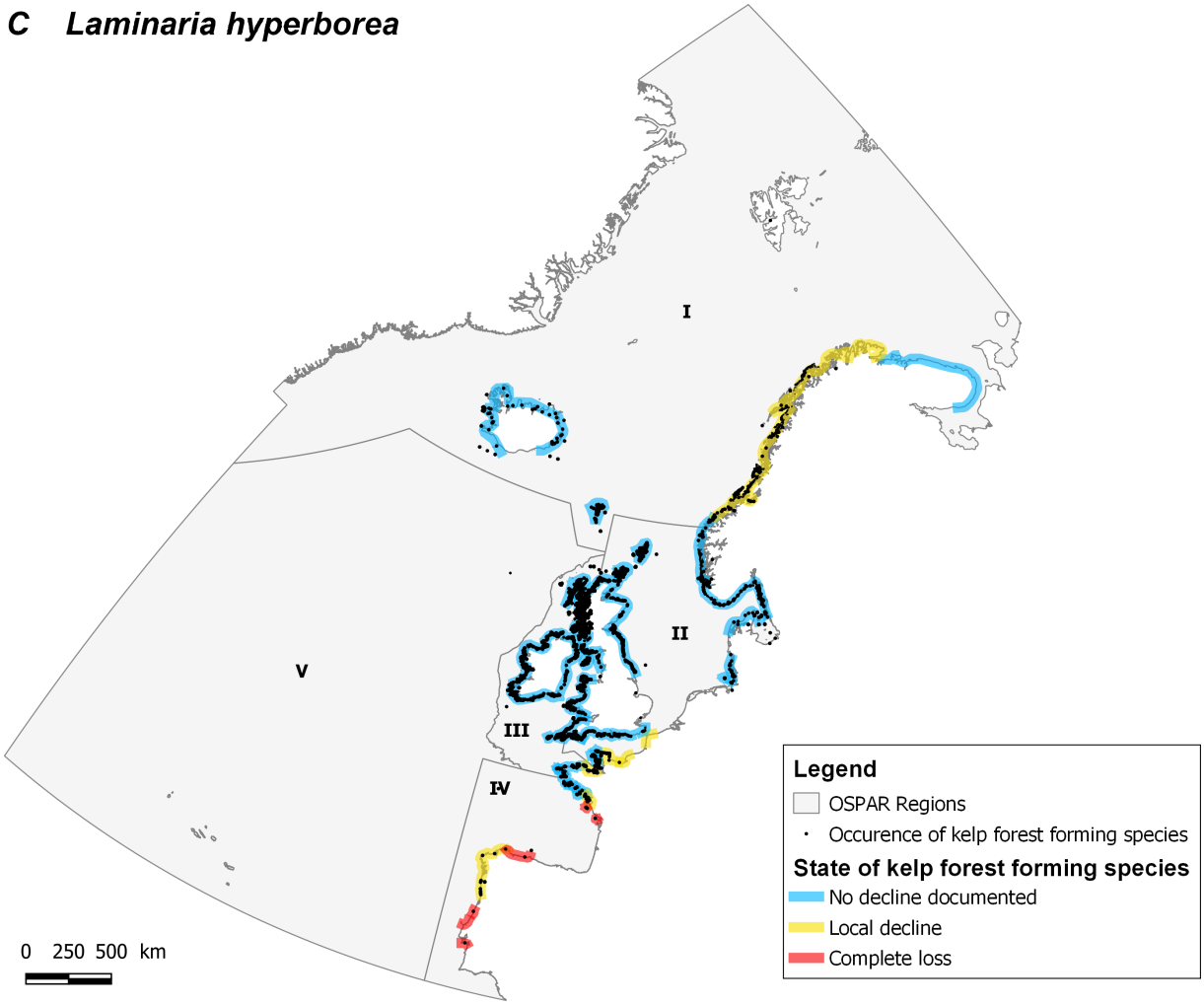
A *Alaria esculenta*



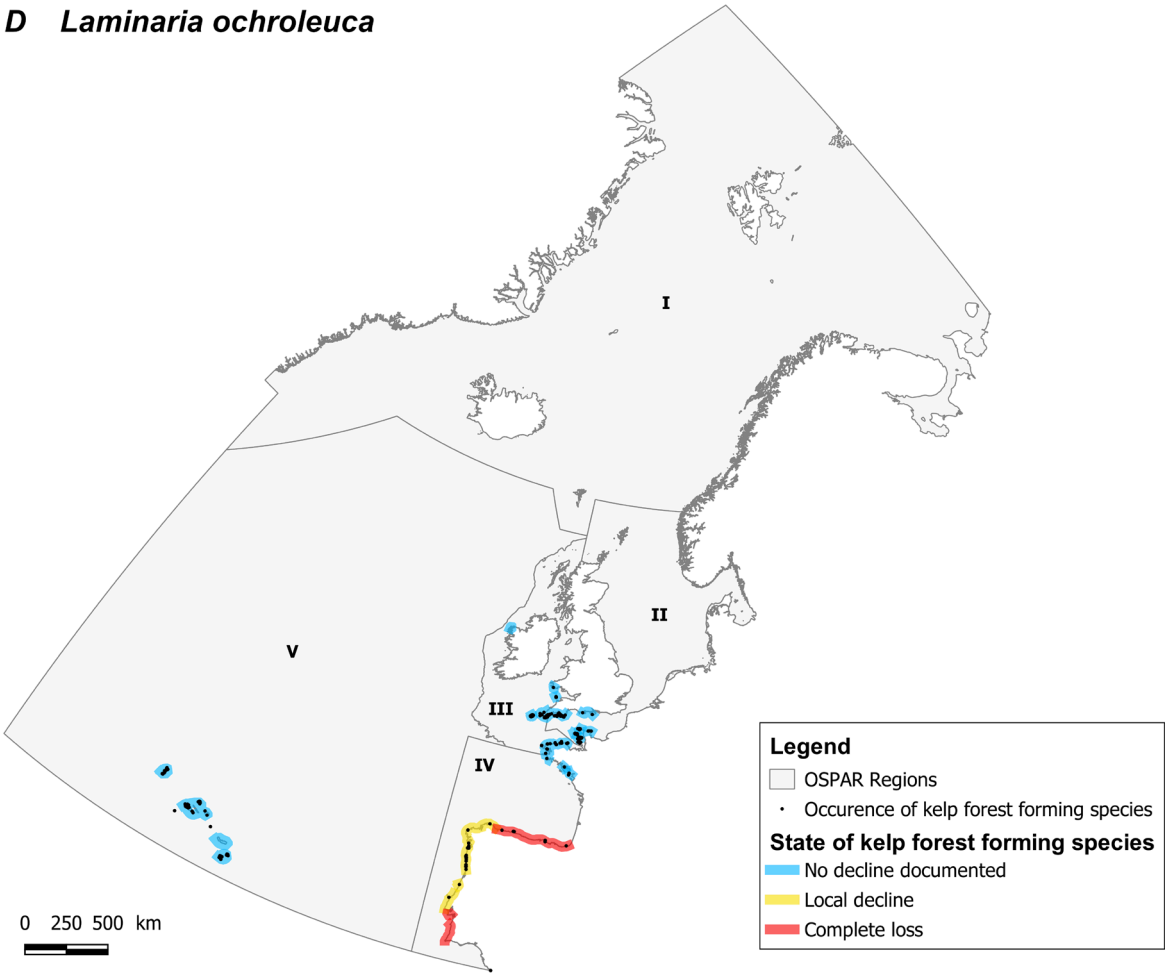
B Laminaria digitata



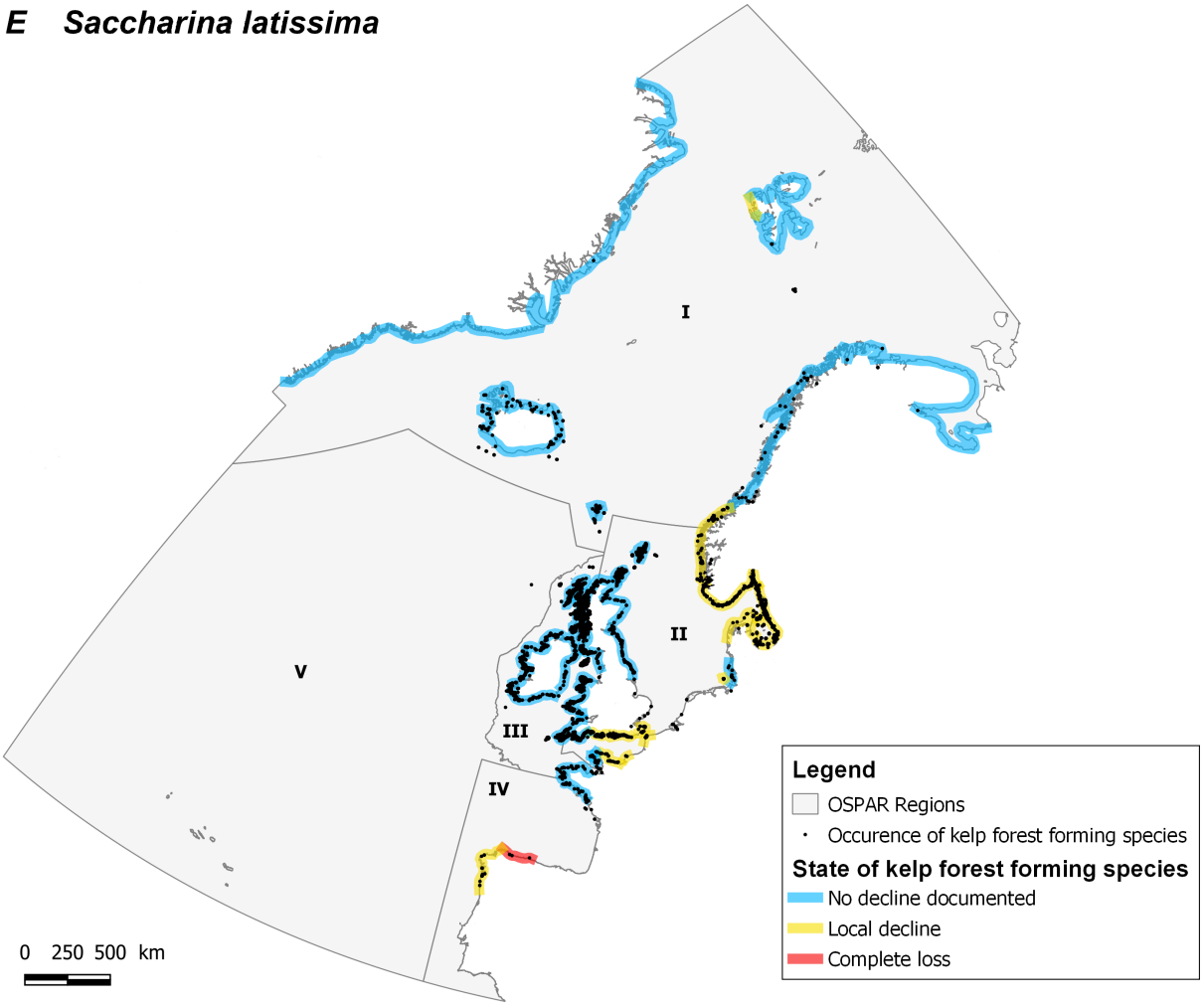
C *Laminaria hyperborea*



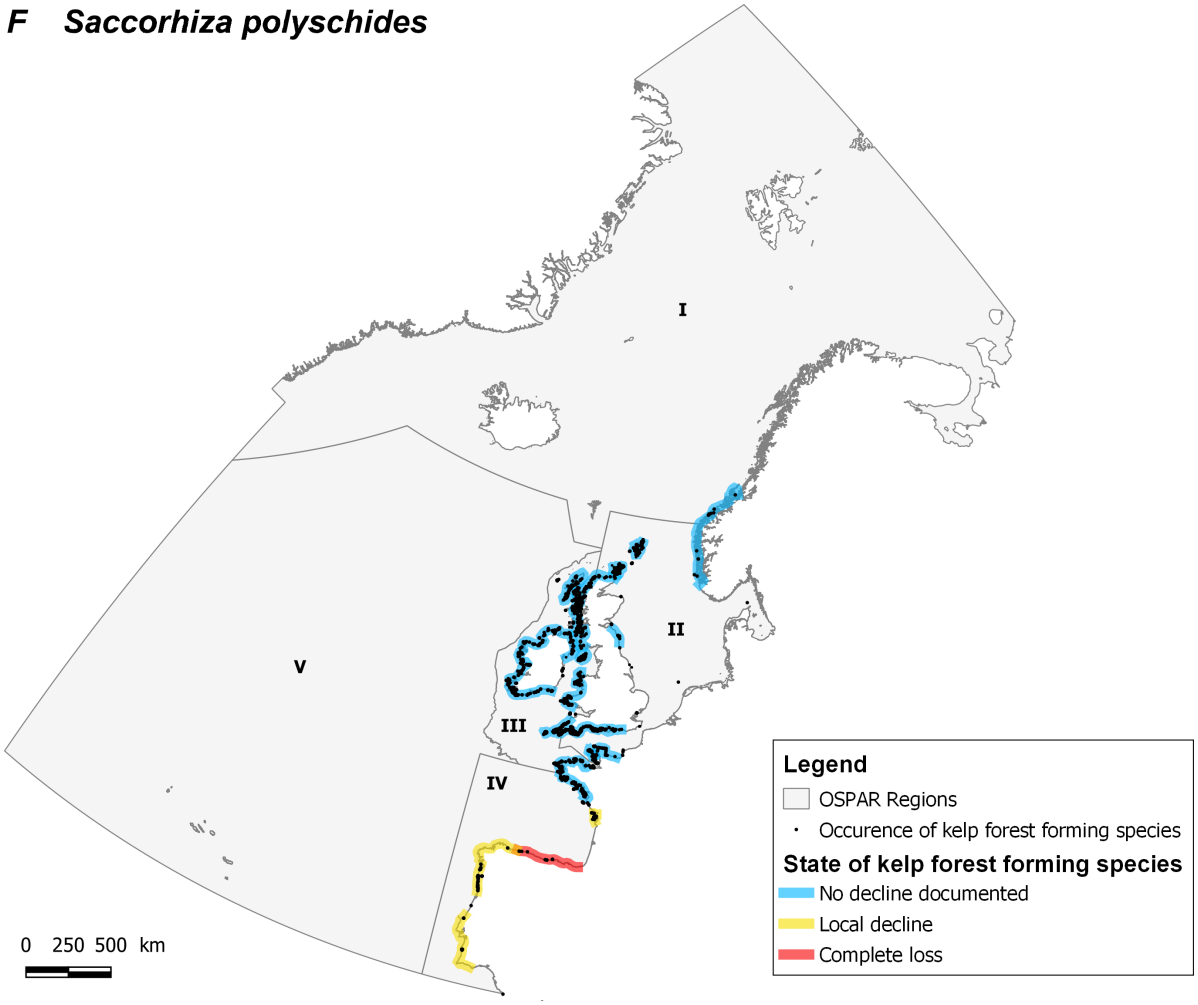
D Laminaria ochroleuca



E Saccharina latissima



F Saccorhiza polyschides



c. Rarity

No

Kelp forest habitat is not rare in the overall OSPAR area. Some species that are dominant in certain OSPAR Regions are patchy or scarcely documented for others (e.g., *S. latissima* is rare on the northwest coast of Spain). In the Azores and the seamounts off southwestern Iberia, the ecological niche of *L. ochroleuca* is naturally constrained by geomorphology and fragmented by their archipelagic geography. However even though kelp forests cannot be considered rare, it is likely that some kelp genotypes are rare and concentrated at the southern edge of their current distribution. If impending genetic analyses confirm population or even species differentiation within *L. ochroleuca*, certain isolated populations would be associated with very limited distribution areas. Similarly, the unique Brittany gene pool of *L. digitata*, at the southern limit of its OSPAR distribution, is projected to disappear in the near future under high greenhouse gas emission scenarios (Neiva *et al.,* 2020).

d. Sensitivity

Sensitive

Kelp forest habitats have a high capacity to resist and recover from natural disturbances, such as wave action and storms (Dayton *et al.*, 1992; de Bettignies *et al.*, 2013 and 2015). However, kelp forest habitats are sensitive to anthropogenic physical, chemical, and biological pressures, with low resistance to warming and vulnerabilities that can lead to dramatic decreases in kelp habitat covering tens to hundreds of kilometres, on time scales as short as one year (Assis *et al.*, 2013; Wernberg *et al.*, 2016). They are considered 'sensitive' although their resistance or resilience varies depending on the dominant species (e.g., *L. digitata* is often less sensitive than *L. hyperborea*), life cycle stages and pressures that they are exposed to. At the edge of their distribution kelp forest habitats exposed to cumulative pressures might have lower resilience (and thus higher sensitivity) than towards the centre of the distribution of these habitats. There are well documented declines in kelp forest abundance and local extinctions at low latitude range limits such as around south-western Australia and the Iberian Peninsula (Wernberg *et al.*, 2010 and 2016; Pereira *et al.*, 2017; Casado-Amezúa *et al.*, 2019). However, in the case of *L. hyperborea* in mid-Norway, close to its northern limit of distribution (southern part of Region I), kelp forest habitats have recovered well after sea urchin overgrazing events (Norderhaug & Christie 2009; Steen *et al*., 2016).

Kelps live in temperate and cold-waters. As a result, kelp forests are very sensitive to an **increase in sea water temperature** (gradual and sudden increases), because higher temperatures in summer as in winter can directly and indirectly lead to kelp decline through a variety of mechanisms (e.g., direct mortality, reduced reproduction, increased herbivory) (Bartsch *et al.*, 2013; Oppliger *et al.*, 2014; Norderhaug *et al.*, 2015a; Martins *et al.*, 2017; de Bettignies *et al.,* 2018; Liesner *et al*., 2020a) and are often associated with lower nutrients (Pereira *et al*., 2015; Assis *et al.,* 2016b). A major threat lies in regions where warming is the most significant (e.g., at the trailing-edge of the species distribution), which in some cases has already led to local extinction (Voerman *et al.*, 2013; Robuchon *et al.*, 2014; Araújo *et al.*, 2016; Assis *et al.*, 2016b; Franco *et al.*, 2018; Casado- Amezúa *et al.*, 2019). The marked distributional contraction of southernmost populations and the low variability in the upper survival threshold to seawater temperature (reviewed In Lüning, 1990; Liesner *et al.,* 2020b) is suggesting reduced adaptability and thereby high sensitivity to increasing seawater temperatures. There is now increasing evidence that discrete temperature events (marine heatwaves) can have a critical impact on kelp and can be the driver factor influencing kelp forest declines in the Atlantic (Filbee-Dexter *et al.*, 2016 and 2020). Such declines have led to phase shifts from luxuriant kelp forests to simplified turf seascapes (Filbee-Dexter & Wernberg, 2018), often associated with range expansion of invasive seaweeds (increased competition) and warm water fishes (increased grazing) (Brodie *et al.*, 2014).

Kelp forests are sensitive to natural or anthropogenic **water quality changes** such as **turbidity, sedimentation and pollution**, and they may be used as bioindicators for the detection of heavy metals and other pollutants

(Derrien-Courtel *et al.*, 2013; Le Gal and Derrien-Courtel, 2015) or the decrease/increase of underwater irradiance (e.g., Bartsch *et al*., 2013 and 2016).

Kelp forests are particularly sensitive to **substratum abrasion** (e.g., by dragging gears) that removes patches of kelp and alters the structure of the habitat**,** especially when the pressure leads to **resuspension of sediments** that can hinder reproduction, recruitment and photosynthesis (Roleda & Dethleff, 2011; Robuchon *et al.*, 2014; Zacher *et al.*, 2016). After removal, the kelp biomass may recover in 1 to 6 years, or faster if propagules are already present (Barradas *et al.,* 2011; Bartsch *et al.*, 2013), while the rich associated community of algae and fauna may take 7 to 10 years to recover completely, if environmental conditions enable the settlement of new recruits (Kain, 1975; Hawkins & Harkin, 1985; Rinde *et al.*, 1992; Birkett *et al.*, 1998; Christie *et al.*, 1998; Davoult *et al.*, 2011; Engelen *et al.*, 2011; Steen *et al.*, 2016; Pereira *et al.*, 2017).

Kelp forests are also sensitive to **removal of target or non-target species** and other pressures that can **alter the ecological food webs** and lead to a regime shift into another stable habitat state that is difficult to recover from. Under sheltered and medium exposed conditions, dense populations of grazers can feed on kelp and later prevent kelp forests from recovering (Ling *et al.*, 2015). A key example is sea urchin populations, which can increase when their predators are overfished and lead to the transformation of former kelp foresthabitats to sea urchin dominated barren grounds (Norderhaug & Christie 2009; Filbee-Dexter & Scheibling, 2014; Ling *et al.*, 2015). The result of such urchin grazing can be low productivity marine deserts, with kelp forests often only prevailing on the most wave exposed outer coastal fringe. Similarly, warm water herbivorous fish prevalent in Portugal and Northern Spain are also likely to damage kelp forests (Franco *et al.*, 2017; Casado-Amezúa *et al.*, 2019), a common and on-going process related to warming and so-called tropicalization of temperate kelp forest ecosystems (Vergés *et al.*, 2014; Wernberg *et al.*, 2016). Another detrimental stable state has been observed in southern Norway, Spain and Portugal with the replacement of collapsed kelp forests by low productive turf seascapes (Filbee-Dexter & Wernberg, 2018; Filbee-Dexter *et al.*, 2020).

e. Ecological significance

Kelp forests support high rates of primary production and create habitat for a multitude of marine organisms. Kelp forests are one of the most productive habitats on Earth with up to 1500-2000 g C m^{-2} yr^{-1} (Mann, 1973; Fredriksen, 2003; Krumhansl & Scheibling, 2012). Within the coastal euphotic zone, kelps produce nearly 75% of the net carbon fixed annually, providing approximately 40% of the available substratum for macroalgal attachment (Birkett *et al.*, 1998). They are the major primary producers of most coastal habitats in the North-East Atlantic. They are 'foundation species' that enhance diversity and secondary productivity at the local scale through the formation of biogenic habitats (*sensu* Dayton, 1972) and at broader scales via detrital subsidies (Krumhansl & Scheibling, 2012). Kelp forests also play a key role in the capture and export of carbon within the food web (Dayton, 1985; Duggins *et al.,* 1989; Krumhansl & Scheibling, 2012; de Bettignies *et al.*, 2013; Leclerc *et al.*, 2013; Feehan *et al*., 2018; Vilas *et al*., 2020). Fundamental change in food web structure induced by changes in kelp habitat following Artic warming and reduced sea ice cover has been documented in Svalbard (Paar *et al.,* 2016 and 2019).

Kelps create habitats (Thomsen *et al.*, 2010) that enable other species to occur among and underneath their canopy, stipes and holdfasts. They provide habitat for sessile flora and fauna, which at the same time, support other fauna such as mobile invertebrates (gastropod molluscs, crustaceans, and echinoderms), some of which have significant ecological or socioeconomic importance (e.g., sea urchins and the European lobster) (Robuchon *et al.*, 2015; Leclerc *et al.*, 2016; Teagle *et al.,* 2018). Therefore, shifts in relative abundance of North-East Atlantic kelp species is forecast to alter local biodiversity patterns and ecological processes (Brodie *et al.*, 2014; Smale *et al.*, 2015). Furthermore, kelp forests ameliorate physical stress (e.g. hydrodynamic forces, UV, pH and to some extent temperature) (Gaylord *et al.,* 2007; Bennett *et al.,* 2015; Krause-Jensen *et al.,* 2016) and can mitigate the effect of climate change stressors on associated stress-sensitive species (Bulleri *et al.*, 2018).

Kelp forests are nursery grounds for juvenile invertebrates and fish sustaining many key and commercially important species by providing shelter from predation such as *Gadus morhua* (cod) and *Pollachius virens* (pollock) (e.g., Brand & Fischer, 2016). They are also significant feeding grounds for many NE Atlantic fish, such as *Labrus bergylta* (Ballan wrasse) and *Ctenolabrus rupestris* (Goldsinny wrasse), which prey on kelp-associated invertebrates (Norderhaug *et al.*, 2005). As a result, these fish occur in high densities within kelp forests, which in turn attract large predators, such as piscivorous fish, seals and other mammals.

Kelp forest habitats store carbon, and play a role in global carbon cycling (Chung *et al.,* 2013; Krause-Jensen & Duarte, 2016; Pessarrodona *et al.*, 2018; Filbee-Dexter & Wernberg, 2020). They can reduce the likelihood of marine flooding and coastal erosion by attenuating wave energy, serving as a buffer during storm events (Løväs & Tørum, 2001), although this is not always the case (Morris *et al*., 2019). This is an important ecosystem service, particularly given predictions related to climate change (e.g., sea-level rise and an increased magnitude and frequency of storms).

The value of the many ecosystem services provided by kelp forests is estimated to be billions of dollars annually (Beaumont *et al.*, 2008; Bennett *et al.*, 2016; Blamey & Bolton, 2018). These include creating nursery and/or feeding grounds to a vast number of species including commercial species, nutrient uptake and shoreline protection (Sjøtun *et al.*, 1995; Jørgensen & Christie, 2003; Abdullah & Fredriksen, 2004; Schaal *et al.*, 2010; Smale *et al.*, 2013, Bertocci *et al.,* 2015; Blamey & Bolton, 2018).

f. Status of decline

Globally kelp forest habitats have been in decline in some areas and on the increase in others (Araujo *et al.,* 2016). Declines have been documented in low latitude distribution limits, while abundances have been stable or increasing in other areas (Araujo *et al.,* 2016; Krumhansl *et al.*, 2016; Wernberg *et al.*, 2019).

In warmer parts of the North-East Atlantic, an overall trend of a decrease in kelp forest distribution and abundance (extent and quality) has been documented (Araújo *et al.*, 2016; Casado-Amezúa *et al.*, 2019). In the southernmost areas, there is evidence of a marked distributional contraction and declines in abundance and vital traits related to global warming (Casado-Amezúa *et al.*, 2019 and references therein). Due to the east to west orientation of the northern coast of the Iberian Peninsula, centred in a short latitudinal range, the contraction in distribution in this area has been synchronic and large. Changes in distribution in this southern limit anticipate predicted responses to the projected warming in the central geographic distribution of the species. Modelling studies project severe declines of kelp forest habitats along much of the European coastline, and a range shift towards cold temperature polar areas (Müller *et al.*, 2009; Krause-Jensen *et al.*, 2012; Assis *et al.*, 2013; Raybaud *et al.*, 2013; Brodie *et al.,* 2014).

In some parts of the OSPAR region kelp forests are recovering, for example there has been a documented increase of *L. hyperborea* along a 500 km coastline of northern Norway over the last 30 years due to reduced sea urchin grazing (Norderhaug & Christie 2009, Rinde *et al.*, 2014). Exceptions to an overall declining trend in kelp forest habitat have been noted in Brittany (France), the English Channel (UK), and the North Sea (Germany). These locations may represent potential refuge areas for conservation although in some cases, kelp populations have not fully recovered to historical levels (Araújo *et al.*, 2016).

In some cases, certain kelp species have proliferated at the expense of others (e.g., *L. hyperborea* increasing at Helgoland, Germany, while *L. digitata* and *S. latissima* have declined) (Pehlke & Bartsch, 2008) and the observed local increase of warm-water kelp species, such as *L. ochroleuca* in the English Channel at its poleward range edge which is now competing with *L. hyperborea* (Smale *et al.*, 2015). The broader ecosystem impacts of these species replacements and proliferation are still not well understood (Pessarrodona *et al.*, 2019).

For some types of kelp forests in certain areas, information gaps and unclear population trends make it difficult to determine whether those kelp forests are threatened and/or in decline. It is notably the case in Ireland where evidence is currently insufficient regarding kelp forests status.

On the basis of the best available information and scientific expertise, the categorisation of decline for each of the six types of kelp forests at the regional level is described below, according to OSPAR Agreement 2019-03 (selection criteria for habitats).

Information provided below is restricted to the kelp forest forming species and the regions where it should be considered a status of decline or probability of significant decline[2]. Information for regions where kelp forest forming species occur but where there is no sign of decline or where the status of decline is unknown or under recovery was documented (BDC 20/7/1 Annex 11).

Alaria esculenta dominated forest

Region II – Probability of significant decline

The evidence shows a mixed picture for this species with some recorded declines and some increases. There was a lack of large-scale consistent monitoring to draw a firm conclusion. The assessment 'probability of significant decline' was made due to predictions of declines in the south due to climate change along with some recorded declines at particular sites.

Evidence shows small-scale variations in trends for some isolated locations but the available records refer mainly to presence/absence data. Yesson *et al.* (2015) reviewed change in historic datasets from UK waters (mostly Region III) and found 37% of sites showed trends of decline and 58% showed trends of expansion in the 1974-2010 period; however, the authors acknowledge that these were not systematic repeated studies and the ad hoc nature of sampling can greatly limit conclusions that can be drawn. Predictive modelling using sea surface temperature suggests that climate change will cause *Alaria esculenta* to retreat from the coasts of south-west England (Mieszkowska *et al.,* 2005). *Alaria esculenta* is particularly susceptible to climate fluctuations, disappearing from much of the English Channel during a warm period in the 1950s and not recovering as conditions became cooler in the 1960s (Southward *et al.*, 1995). Temperature affinities of *A. esculenta* are rather cold-temperate to Arctic as gametogenesis and sporophyte recruitment are fully suppressed over 15°C (Zacher *et al.*, 2019). The same might be true for spore formation of the summer-fertile sporophyte. The temperature limit of this process is probably even lower than the gametogenesis temperature (according to concept of Bartsch *et al.*, 2013). Thus, this species may probably only survive in cool-water pockets in the south allowing the completion of the life-cycle and this will become more restricted as temperature increases.

Region III – Probability of significant decline in the French EEZ

The evidence shows a mixed picture for this species across much of the region with some recorded declines and some increases. The assessment 'probability of significant decline' was made due to predictions of declines in the south due to climate change along with some recorded declines in Brittany.

Yesson *et al.* (2015) reviewed change in historic datasets from UK waters (mostly Region III), and found 37% of sites showed trends of decline and 58% showed trends of expansion in the 1974-2010 period (with a possible decrease in the Irish Sea and no clear trend in other areas). However, the authors acknowledge that these were not systematic repeated studies and the ad hoc nature of sampling can greatly limit conclusions that can be drawn. Predictive modelling using sea surface temperature suggests that climate change will

[2] 'Probability of significant decline' has been assessed by expert judgement and has been applied following Agreement 2019-03, where that there is a high probability that the habitat will decline by 15% or more in extent, or that the habitat will significantly decline in quality if no protection or management measures are taken. OSPAR Workshop on draft Case Report for listing Kelp Forests concluded that features meeting the criteria 'Probability of significant decline' should be proposed for nomination on the OSPAR list.

cause *Alaria esculenta* to retreat from the coasts of south-west England (Mieszkowska *et al.,* 2005). In France, it is distributed between Brittany (Pointe du Van, South Iroise) and Normandy (Cherbourg, region II), with optimal development in Iroise (west of Brittany) (MEDDE, 2012). The species has completely disappeared from southern Brittany (Plogoff 48°N, southern tip of Region III) where it was still observed 20 years ago (Castric-Fey *et al.*, 2001). Temperature affinities of *A. esculenta* are rather cold-temperate to Arctic as gametogenesis and sporophyte recruitment are fully suppressed over 15°C (Zacher *et al.*, 2019). The same might be true for spore formation of the summer-fertile sporophyte. The temperature limit of this process is probably even lower than the gametogenesis temperature (according to *Bartsch et al.*, 2013). Thus, this species may probably only survive in cool-water pockets in the south allowing the completion of the life-cycle and this will become more restricted as temperature increases. The cooler temperature required for zoospores (upper limit of 18°C and optimal germination between 2-12°C) may restrict this species in the region if temperature continue to increase (Yesson *et al.*, 2015). Some significant signs of decline have been observed in Ireland between 1958 and 2003 (Simkanin *et al.*, 2005).

Laminaria digitata dominated forest

Region II – Probability of significant decline

The evidence shows a mixed picture for this species across much of the region with some recorded declines and some increases. The assessment 'probability of significant decline' was made due to predictions of declines in the south due to climate change along with some recorded declines along the French coast.

Yesson *et al.* (2015) reviewed change in historic datasets from UK waters, and found 34% of sites showed trends of decline and 64% showed trends of expansion in the 1974-2010 period with more declines on the south coast and increases in the north; however, the authors acknowledge that these were not systematic repeated studies and the ad hoc nature of sampling can greatly limit conclusions that can be drawn. Populations of *L. digitata* in the eastern English Channel and Dover Strait (51°N) along the French coast are now extinct (Araújo *et al.,* 2016). Further west, close to the region II and III boundary, forests of *L. digitata* progressively disappeared (1983/1988-1997) on the coast of Lower Normandy (49°N) with an estimated loss of ~ 88 000 Tones (Cosson*,* 1999). A majority of these forests are now replaced by an invasive macroalgal species, *Sargassum muticum* (Cosson*,* 1999). In the North Sea, these forests are under important pressure and reductions in abundance and bathymetric distribution were observed (Bartsch *et al.,* 2013). At the island of Helgoland (54°N) in the North Sea, this species significantly declined between 1968 and 2005 (Pehlke & Bartsch, 2008). *L digitata* is also found on the open reefs in Kattegat with varying densities (Dahl, pers. com.).

Region III – Probability of significant decline in the French EEZ

The evidence shows a mixed picture for this species across much of the region with some recorded declines and some increases. The assessment 'probability of significant decline' was made due to predictions of declines in the south due to climate change along with some recorded declines along the French coast.

The highest abundance of *L. digitata* is found in western and north-western regions of French Brittany (entire French part of region III) and correlates with higher genetic diversity (Billot *et al.*, 2003; Valero *et al.*, 2011; Couceiro *et al.*, 2013; Robuchon *et al.*, 2014). The results suggest that the southernmost populations of *L. digitata*, located in Northern France, particularly small and isolated populations in Brittany, have diminished in recent decades leading to a poleward contraction of the trailing range edge (Raybaud *et al.*, 2013; Oppliger *et al.,* 2014). While no extensive data are supporting other local declines in Brittany, the habitat discontinuities in Northern Brittany (i.e., long sandy beaches) led to the isolation of small kelp populations with similar poor genetic and demographic characteristics (reduced genetic diversity, low immigration and reduced population size) than range edge populations in Southern Brittany (Valero *et al.*, 2011; Oppliger *et al.*, 2014). This habitat discontinuity impacts *L. digitata* forests from Locquirec (48°N, 3°W) to Saint-Malo (48°N, 1°W). Similarly to range limit kelp populations more prone to local declines, such marginality through fragmentation by distance (Valero *et al.*, 2011; Oppliger *et al.*, 2014) can lead to increasing declines in this

region. Nevertheless, kelp populations are genetically highly diverse and well connected in the Iroise Sea (48°N, 4°W) where it is exploited since the beginning of the 19th century for its alginates (Valero *et al.*, 2011; Couceiro *et al.*, 2013; Derrien-Courtel, pers. com.). Yesson *et al.* (2015) reviewed change in historic datasets from UK waters and found 34% of sites showed trends of decline and 64% showed trends of expansion in the 1974-2010 period (with a possible decrease in the Irish Sea and no clear trend in other areas); however, the authors acknowledge that these were not systematic repeated studies and the ad hoc nature of sampling can greatly limit conclusions that can be drawn. While some significant signs of decline have been observed in Ireland between 1958 and 2003 (Simkanin *et al.*, 2005), there is currently insufficient scientific evidence to robustly determine species' trends and it cannot definitively be stated that any kelp forest-forming species is currently in decline.

Region IV – Significantly declined

Laminaria digitata forests are restricted in region IV to a small portion of the French coast, from southern Brittany to the Ile d'Oléron (45°55'N), that marks its southern limit. The species is in strong decline on the southern Brittany coasts, bathed by stratified waters and it has disappeared since the mid-1980s in Oléron, 45°N (MEDDE, 2012).

Laminaria hyperborea dominated forest

Region IV – Severely declined

This species has drastically declined in region IV in extent where *L. hyperborea* forests have declined over more than 900 km of coastline (Fig. 2) and in quality where even local losses of kelps lead to drastic changes in the overall kelp forests ecosystem functioning at the regional level. In Spanish and Portuguese waters, a complete loss of habitat has led to a range contraction of 14% (Northward shift of 72 km², westward shift of 118 km²), in the last ca. 30 years. In the Cantabrian Sea (Bay of Biscay), more than 90% of the *L. hyperborea* forests have disappeared, and extinction of populations in the lower intertidal zone in the Rías Altas and Rías Bajas in the western Galician region have been observed (decrease by 83% between 1998 and 2014) (Casado-Amezúa *et al.*, 2019; and references therein). This led to a dual contraction of 150 km from the Bay of Biscay westward along northern Spain and 250 km northward from the previous southern range limit (41°N, Northern Portugal; Assis *et al.,* 2009). Some isolated and small sized populations remain in Northern Portugal and Galicia. This species has also been reported to have shifted into deeper waters (Martinez *et al.*, 2015). French experts reported a decreasing trend or stability of *L. hyperborea* in Brittany (Araújo *et al.,* 2016). This species colonizes only very occasionally the coasts located south of the Gironde estuary in France (45°N). When present, it develops essentially in depth under the thermocline. Above the thermocline, the decline of *Laminaria hyperborea* favours the extension of *Saccorhiza polyschides* (MEDDE, 2012). In southern Brittany, *S. polyschides* seems to replace gradually *L. digitata* and *L. hyperborea*; a phenomenon likely to be facilitated by increasing disturbance (abrasion and removal of species due to commercial harvesting and storms) (Arzvel, 1998; Engelen *et al.*, 2011; Derrien-Courtel pers. com.). Future projections predict a 50% decline of the potential area of *L. hyperborea* in the 40 years to come, particularly under more negative greenhouse gas emissions scenarios (Assis *et al.*, 2016b and 2018a).

Laminaria ochroleuca dominated forest

Region IV – Severely declined

Data on temporal trends of *L. ochroleuca* indicate a general massive reduction in population abundance, where the forests have already declined from more than 94% of the known distribution (Fig. 2) (Araújo *et al.*,

2016; Casado-Amezúa *et al.*, 2019). This species has drastically declined in the Spanish and Portuguese waters where it was the dominant species of the subtidal zone at many sites covering over 75% of the seabed. The complete loss of habitat has led to a westward range contraction of 10% westwards (144 km²) in the last ca. 30 years (Casado- Amezúa *et al.*, 2019; and references therein). Since the 1980's, *L. ochroleuca* populations have been observed to shift to the western part of the Spanish Atlantic coast, as the warm-water enclave of the southern Bay of Biscay expands and the colder waters of northwestern Iberia provide a temporary and contracting refuge. Some populations remain in intertidal to shallow subtidal sites in northern Portugal and in deeper waters further south (Assis *et al.,* 2018a, b; Pereira *et al.*, 2019). Former common and dense populations of *L. ochroleuca* have been reduced to scattered individuals and small patches East of Cap de Peñas in Asturias (Fernandez, 2011; Martinez *et al.,* 2015). Similar trends have also been detected on the northern coast of France, while other populations of this species have increased or stabilised in recent years (Araújo *et al.*, 2016). *L. ochroleuca* has been recorded locally in Loire Atlantique where an increase of its occurrence and abundance is expected (Derrien-Courtel, pers. com.). The species used to be abundant on the Arrabida coast, its recent southern limit in mainland Portugal, where it could not be found anymore in 2018 (project INFORBIOMARES monitoring). In the 1980's at least one herbarium specimen was collected much further south on the Alentejo coast (Serraõ, pers. com.; Fig. 2), a few hundred kilometres south of its southernmost limit on the mainland.

Along the Portuguese mainland coast, isothermal submergence has been noted in the occurrences: while populations in the north occur in intertidal to shallow subtidal sites, records towards southern Portugal tend to be in deeper waters (Assis et al. 2018a, b; Pereira *et al.*, 2019). In the 1980's at least one herbarium specimen was collected on the Alentejo coast (Serrão, pers. com.; Fig. 2), a few hundred kilometres south of its current southernmost limit on mainland Portugal. The species also used to be abundant on the Arrábida southern coast, where its distribution has become restricted over time (program BIOMARES monitoring) and is at the present southern limit of this species in continental Portugal. The high genetic richness of these southern edge populations between Cascais and Arrábida indicates that this marginal refugium has been persistent for longer than all existing populations further north, which are more recent colonisations (Assis *et al.,* 2018b).

No time series exists for the abundance of this species on the Gorringe seamount (southwestern Iberia, offshore Region IV) where the species occurs between 32 and 88 m depth (Ramos *et al.,* 2016). The remote and oceanic character of the few euphotic seamount summits off southwestern Iberia is likely to offer some buffering from natural and anthropogenic surface stressors. Projections by Assis *et al.* (2018a) estimate that the thermal niche of *L. ochroleuca* in these locations is expected to grow between 49 and 16% by the 2090-2100 decade, depending on the emission scenario retained (RCP 2.6 or RCP 8.5, respectively). These offshore sites may therefore play a role as climatic refuges for certain kelp species and their genetic diversity, which has been brought to light by Assis *et al.* (2016a, b).

Saccharina latissima dominated forest

Region II – Significantly declined

In the most Northern part of Region II *S. latissima* has suffered from overgrazing as detailed in the previous section for Region I. Forests of *S. latissima* were replaced by silty turf communities dominated by filamentous opportunistic ephemeral algae, resulting in a much lower species richness and abundance (Christie *et al.*, 2009; Filbee-Dexter *et al.*, 2020). The reduction in abundance of *S. latissima* has been estimated to be 50-90% in the Norwegian Skagerrak (Bekkby & Moy 2011; Moy & Christie, 2012), 40-50% in the Norwegian North Sea coast (Moy & Christie, 2012; Lindgaard & Henrikson, 2011). Yesson *et al*. (2015) reviewed change in historic datasets from UK waters and found 39% of sites showed trends of decline and 58% showed trends of expansion in the 1974-2010 period; however, the authors acknowledge that these were not systematic repeated studies and the ad hoc nature of sampling can greatly limit conclusions that can be drawn. *Saccharina latissima* plants may become smaller and less productive in UK waters as the ocean warms (Pessarrodona *et al.,* 2018). *S. latissima* disappeared or became less abundant on transects investigated in

several Danish fjords bordering the North Sea and Kattegat in 1993 and 1994 and the stock has not fully recovered since (Dahl, pers. com.). On boulder reefs in the open part of Kattegat no decline in the stocks of *S. latissima* has been observed (Dahl *et al.*, 2013). In south-western Norway, the decreasing trend was less severe especially in the most exposed areas, although large sectors were lost in sheltered areas (Moy & Christie 2012; Norderhaug *et al.*, 2015a and b). Since 2005, while the *S. latissima* kelp forests have marginally improved, large areas are still degraded (Moy *et al.*, 2015; Norderhaug *et al.*, 2015a). Following this decline in Norway, temporal and spatial variations in abundance and cover suggest a plausible natural fluctuation between seabed states (i.e., sugar kelp forests and turf algae mats) (Christie *et al.*, 2019). However, the use of coarse quality abundance scale in Christie *et al.* (2019) can be problematic to adequately detect variation at this level and recent paper by Filbee-Dexter *et al.* (2020) highlighted the expansion and persistence of turf seascapes likely to be driven by warm summers (marine heatwaves) in southern Norway.

Overall, available data show a dramatic reduction in abundance, biomass, cover and/or depth of occurrence, and extinction of *S. latissima* in large and multiple areas of Region II (e.g., eastern English Channel, Dover Strait, Helgoland, Sweden, Denmark and Germany coasts) (Cosson, 1999; Eriksson *et al.*, 2002; Pehlke & Bartsch, 2008; Bekkby & Moy, 2011; Araújo *et al.*, 2016).

Skagerrak's *S. latissima* forests are listed as Endangered in the Norwegian Red List, with criteria 1.2 (i.e., for habitats with substantially reduced surface area: reduction of the area of occupancy in the last 50 years = 50-80%). North Sea *S. latissima* forests are listed as Vulnerable on the Norwegian Red List for habitats, with criteria 1.2 (i.e., for habitats with substantially reduced surface area: reduction of the area of occupancy in the last 50 years = 30-50%). *S. latissima* forests have also been upgraded to Endangered (due to their expected continuing degradation for the next fifty years) in the most recent Norwegian Red List for Ecosystems and Habitat Types.

Region IV – Significantly declined

Since 2000, the decline of intertidal and shallow subtidal populations has been documented in the Iberian Peninsula and this species is now almost extinct in this area, being restricted to the Rías Altas and Rías Bajas where forests shifted to deeper waters (Martinez *et al.,* 2015; Casado-Amezúa *et al.,* 2019). In the last ca. 30 years, this reduction has led to a northward and westward contraction of 51 km² in total (Araújo *et al.,* 2016; Casado-Amezúa *et al.,* 2019 and references therein). In some cases (e.g., Portugal), the reduction in biomass has been associated with a shift in depth distribution, with the species now being absent from the low intertidal and restricted to deeper ranges (Martinez *et al.*, 2015; Araújo *et al.*, 2016).

Future projections predict a total extinction from the Iberian Peninsula before 2080 (Assis *et al.*, 2018a), corresponding to a 14% decline of the population in the next 10 years (Casado- Amezúa *et al.*, 2019). All extant populations of *S. latissima* in Portugal and Spain were extensively surveyed and sampled by scuba-diving by Neiva *et al.* (2018). These populations were a genetically distinct clade, isolated from populations of Brittany and containing a rich unique gene pool, which is now vanishing (Assis *et al.*, 2018a; Neiva *et al.*, 2018).

Saccorhiza polyschides dominated forest

Region IV - Significantly declined

S. polyschides is one of the most important species characterizing the low intertidal and shallow subtidal coasts in southern Europe, and the only large kelp species reaching the south-western Algarve coasts of Portugal (Assis *et al.*, 2009). This species has severely decreased in abundance over the last 4 decades (Fernandez, 2011 ; Assis *et al.*, 2013). Former common and dense populations have shifted from the east to the western part of the Spanish coast and northwards from southern Portugal, and to deeper waters. They are now being restricted to forests in the Rías Altas and Rías Bajas and only to scattered individuals and small

patches in the East of Cap Peñas (43°N). The overall contraction range with complete habitat loss is estimated to 172 km^2 (Casado-Amezúa *et al.*, 2019; and references therein). The long-term study of populations showed a severe reduction in *S. polyschides* performance (lower growth of the sporophyte and recruitment) from the early 2000's (Fernandez, 2011) and earlier (Assis *et al.,* 2013). In the northern Iberian Peninsula (with the exception of the Bay of Biscay), this species seems to have maintained a stable distribution over the last years. In the south, two sharp declines in density occurred from 1960 to nowadays (Assis *et al.*, 2013). These declines were also accompanied by strong demographic regressions (decrease in density and increase in fragmentation), local extinctions, and extinctions followed by recolonizations that were confirmed by genetic analyses (Assis *et al.*, 2013). A trend of decrease in *S. polyschides* has been highlighted for some localities of France where it is present from the Southern Brittany coasts to the North of the region Nouvelle-Aquitaine (MEDDE, 2012; Araújo *et al.*, 2016). Southernmost populations and deep refugia have been persistent in the long-term, as indicated by their genetic richness (Assis *et al.,* 2016a). Its predicted distributional changes might not be impaired by dispersal in space or time as demonstrated by dispersal simulations at its southern range limits (Assis *et al.*, 2017).

f) Relevant additional considerations

a. Sufficiency of data

Kelp forests have been well studied for over 40 years in the North-East Atlantic, both in general and in particular locations, through biological and socioeconomic studies. However, kelp forests monitoring has been too short to fully understand changes in relation to background variations (Krumhansl *et al.*, 2016) and long-term monitoring are still strongly needed. The kelp forests of East Greenland have just been recently explored (Wegeberg *et al.*, subm.). There is increasing evidence that rapid changes occur in North-East Atlantic kelp forests in response to climate change (Smale *et al.*, 2013; Brodie *et al.,* 2014; Araújo *et al.*, 2016; Casado-Amezúa *et al.*, 2019 ; Filbee-Dexter *et al.*, 2020). Despite this, there are still considerable knowledge gaps regarding the nature and degree of the alterations caused in these ecosystems by means of climate change, overgrazing, chronic eutrophication, overharvesting, etc., and how these factors are likely to interact and may affect kelp forests resilience (Smale *et al.*, 2013). Trophic interactions in kelp forests (e.g., Leclerc *et al.*, 2013; Paar *et al.*, 2016 and 2019) and kelp-kelp species interactions (e.g. Zacher *et al.,* 2019), especially under global change scenarios, also remain poorly understood and targeted research is required to address the level of threat posed by top-down processes. Specific research programs will also have to be carried out on 'pressure – impact' relationships, in order to be able to measure, quantify and differentiate the impacts of the dominating anthropogenic pressures.

b. Changes in relation to natural variability

The distribution of kelp forests can vary because of variation in natural parameters such as nutrient availability, light penetration, temperature, wave exposure and disturbance regime. These variations can generate contrasting trends (i.e., small-scale spatial variability, with some species increasing in parts of their geographical distribution but decreasing in other areas). In some cases, opposing trends occur a few kilometres apart from each other, which are related to variation in physical conditions in the habitat (e.g., wave exposure, depth, etc.) (Araújo *et al.*, 2016). For example, a correlation seems possible between depth distribution of kelp forests and the NAO index (Derrien-Courtel, pers. com.).

c. Expert judgement

A European-wide expert assessment of the status, trends, and drivers of kelp forests found an overall decreasing trend in kelp forest distribution and abundance across Europe (Araújo *et al.,* 2016). This study also highlighted a lack of long-term, quantitative datasets in large parts of the geographical range, and as a consequence, difficulties in identifying large-scale spatial trends for individual kelp species (Krumhansl *et al.,* 2016). Expert judgement has therefore played a major role in putting forward this nomination to OSPAR. This is because there is a lack of quantitative data for the distribution and abundance trends of certain species, especially in regions I and V, which are most remote. The main conclusion is that despite lack of extensive quantitative and long-term datasets, there is firm evidence that kelp forest habitat is impacted to a variable degree across their distribution and are threatened by various anthropogenic and environmental pressures. They should therefore be listed by OSPAR for each of the regions with nominated kelp forest dominated species, where these declining trends are evident (see Table 1).

g) Trends and prospects on the status of the habitat

Because of the current status of kelp forest habitats and the increasing threat to marine ecosystems posed by global change, management measures, where appropriate, will need to be collectively agreed upon and effectively implemented to reduce the pressures on this habitat and restore its resilience.

Future studies should address the knowledge gaps in kelp distribution (especially in regions I and V), population dynamics and genetic diversity (all regions), resilience and the ecological status and functionality of the kelp habitat, including their role in coastal protection, as carbon storage, as a form of mitigation for species under stress from climate change and as a nursery ground for commercially valuable fish species. Such knowledge would enable a better understanding of the potential effects of pressures and improved management measures. Some locations where increase in forest abundance have been recorded may represent potential refuge areas for conservation, which is not always indicative of a high genetic diversity (Valero *et al.* 2011) and could be one focal area of further investigation.

h) Conclusion on overall status

The kelp forest habitat is characterised by both a high ecological productivity and economic value. Major losses in the southern part of the OSPAR area and significant declines at several locations of the nominated kelp forest habitat have been well documented. The main threats to this habitat are climate change, increased grazing pressures, expansion of algal turfs, reduced water quality and overharvesting, which together may generate synergistic pressures on kelp forests.

The limitations in knowledge described above, for example on the distribution and extent of kelp forests in some areas and the complex ecological dynamics of the forest structure in response to pressures (whether they are natural or anthropogenic) should not prevent the nomination of this habitat type to promote measures to limit further decline.

Table 1 summarises the status of decline for each kelp forests dominating species at the regional level. Information provided in the table is restricted to the kelp forest forming species and the regions where it should be considered as in a status of decline or where there is a probability of significant decline. Information for regions where the species occurs (including other kelp species: *A. clathratum, H. nigripes, L. solidungula, S. dermatodea*), but where there is no sign of decline or where the status of decline is unknown or under recovery, were documented in BDC 20/7/1 Annex 11.

Table 1. Summary of the status of decline at the regional level for the different kelp forests in the North-East Atlantic.

Kelp forests dominating species	Region I	Region II	Region III	Region IV	Region V
A. esculenta	Status of decline unknown	Probability of significant decline	Probability of significant decline in the French EEZ	-	Status of decline unknown
L. digitata	Status of decline unknown	Probability of significant decline	Probability of significant decline in the French EEZ	Significantly declined	Status of decline unknown
L. hyperborea	Not declining (current recovery following a drastic decline)	Not declining (documented regression in France)	Status of decline unknown	Severely declined	Status of decline unknown
L. ochroleuca	-	Not declining	Not declining	Severely declined	Status of decline unknown

S. latissima	Status of decline unknown	Significantly declined	Status of decline unknown	Significantly declined	Status of decline unknown
S. polyschides	Status of decline unknown	Status of decline unknown	Status of decline unknown	Significantly declined	Status of decline unknown

i) Overview of data and information provided by Contracting Parties

Contracting Party	Feature occurs in CP's maritime area	Contribution made to the assessment (e.g. data/information provided)	Names of contributors to the drafting of the document
Belgium	N	N	
Denmark	Y	Y	Dahl Karsten
European Commission			
France	Y	Y	Thibaut de Bettignies, Marie La Rivière, Marine Robuchon, Myriam Valero, Line Le Gall, Sandrine Derrien, Claire Hebert
Germany	Y	Y	Miriam Mueller, Thorsten Werner, Janos Hennicke, Inka Bartsch
Iceland	Y	N	
Ireland	Y	Y	
Netherlands	N	N	
Norway	Y	Y	Karen Filbee-Dexter, Trine Bekkby, Hartvig Christie, Frithjof Moy, Kjell-Magnus Norderhaug, Henning Steen
Portugal	Y	Y	João Franco, Fernando Tempera, Ester Serrão, Jorge Assis
Spain	Y	Y	Brezo Martinez, Sandra Hernández Arenas
Sweden	Y	Y	
United Kingdom	Y	Y	Hugh Edwards, Katie Gillham, Marion Harrald, Nova Mieszkowska, Pippa Moore, Megan Parry, Kirsten Ramsay, Trudy Russell, Dan Smale and Margaret Street

j) References

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Appendix I. Corresponding EUNIS habitats units

25 EUNIS 2012 habitats units that can correspond to the kelp forest habitat as defined in the proforma have been identified by scientific experts based on the EUNIS description and the case report's definition of the habitat (substratum, species and density criteria). For example, habitats units described for soft rocky bottoms, vertical rocky bottoms, Sabellaria reefs, or muddy environment have not been included because the kelp forest habitat (as defined in the proforma) is not believed to be encountered on such substrates. Habitats units corresponding to grazed states of kelp forests have been included.

"Parents" habitats are only cited if all their "children" habitats also correspond to kelp forests as defined in the case report.

EUNIS Level 5 and Level 6 units are indicated in blue and black respectively.

A3.111 *Alaria esculenta* on exposed sublittoral fringe bedrock

A3.1111 *Alaria esculenta*, *Mytilus edulis* and coralline crusts on very exposed sublittoral fringe bedrock

A3.1112 *Alaria esculenta* and *Laminaria digitata* on exposed sublittoral fringe bedrock

A3.112 *Alaria esculenta* forest with dense anemones and crustose sponges on extremely exposed infralittoral bedrock

A3.113 *Laminaria hyperborea* forest with a faunal cushion (sponges and polyclinids) and foliose red seaweeds on very exposed infralittoral rock

A3.1151 *Laminaria hyperborea* forest with dense foliose red seaweeds on exposed upper infralittoral rock

A3.1153 Mixed *Laminaria hyperborea* and *Laminaria ochroleuca* forest on exposed infralittoral rock

A3.121 *Saccorhiza polyschides* and other opportunistic kelps on disturbed upper infralittoral rock

A3.122 *Laminaria saccharina*[3] and/or *Saccorhiza polyschides* on exposed infralittoral rock

A3.125 Mixed kelps with scour-tolerant and opportunistic foliose red seaweeds on scoured or sand-covered infralittoral rock

A3.211 *Laminaria digitata* on moderately exposed sublittoral fringe rock

A3.2111 *Laminaria digitata* on moderately exposed sublittoral fringe bedrock

A3.2112 *Laminaria digitata* and under-boulder fauna on sublittoral fringe boulders

A3.2121 *Laminaria hyperborea* forest, foliose red seaweeds and a diverse fauna on tide-swept upper infralittoral rock

A3.2131 *Laminaria hyperborea* forest and foliose red seaweeds on tide-swept upper infralittoral mixed substrata

[3] Habitats types are transcribed with their official names in the EUNIS 2012 classification which doesn't incorporate more recent taxonomical modification in the species names (e.g. *Laminaria saccharina* changed into *Saccharina latissima*).

A3.2141 *Laminaria hyperborea* forest and foliose red seaweeds on moderately exposed upper infralittoral rock

A3.2143 Grazed *Laminaria hyperborea* forest with coralline crusts on upper infralittoral rock

A3.221 *Laminaria digitata*, ascidians and bryozoans on tide-swept sublittoral fringe rock

A3.222 Mixed kelp with foliose red seaweeds, sponges and ascidians on sheltered tide-swept infralittoral rock

A3.311 Mixed *Laminaria hyperborea* and *Laminaria ochroleuca* forest on moderately exposed or sheltered infralittoral rock

A3.3121 Mixed *Laminaria hyperborea* and *Laminaria saccharina* forest on sheltered upper infralittoral rock

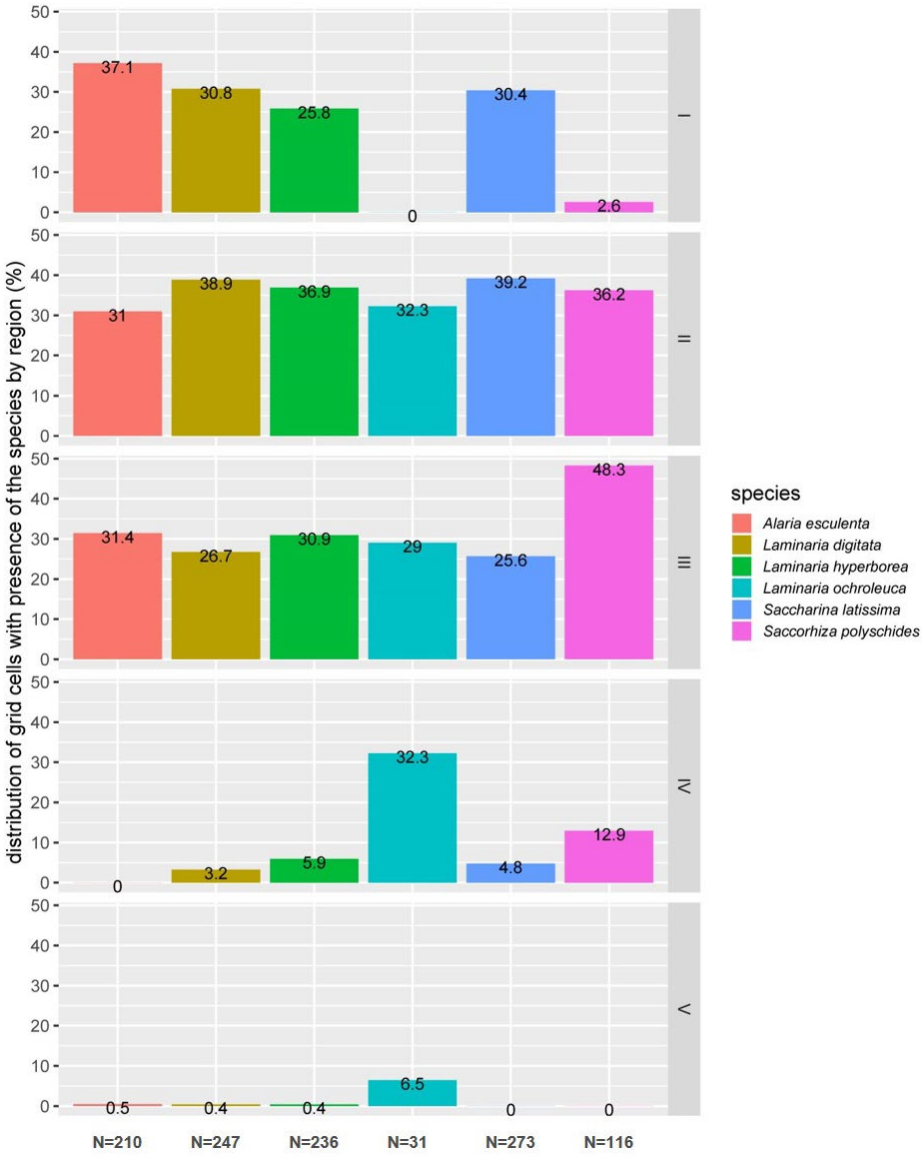
A3.3123 Grazed, mixed *Laminaria hyperborea* and *Laminaria saccharina* on sheltered infralittoral rock

A3.3131 *Laminaria saccharina* and *Laminaria digitata* on sheltered sublittoral fringe rock

A3.3132 *Laminaria saccharina* forest on very sheltered upper infralittoral rock

A5.523 *Laminaria saccharina* with *Psammechinus miliaris* and/or *Modiolus modiolus* on variable salinity infralittoral sediment

Appendix II. Contribution of each OSPAR region to the distribution of six kelp forest habitat-forming species in the OSPAR maritime area



Distribution data are based on data from the Global Biodiversity Information Facility (GBIF, 2019), the Ocean Biogeographic Information System (OBIS, 2019) and data provided by Contracting Parties (UK, Sweden, Norway and France) in 0.5°*0.5° grid cells. N: total number of grid cells with presence of the species in the whole OSPAR area.

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