Cyfoeth Naturiol Cymru
Natural Resources Wales

Hand harvesting of seaweed: Evidence review to support sustainable management

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Author Name: C. Wilding[1], H. Tillin[1], E. J. Stewart[1], M. Burrows[2], D. Smale[1]

Author Affiliation: Marine Biological Association[1], Scottish Association of Marine Sciences[2]

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Comisiynodd Cyfoeth Naturiol Cymru'r adolygiad llenyddiaeth hwn i wella dealltwriaeth o'r ystod o effeithiau posibl casglu gwymon â llaw ac i adolygu mesurau rheoli posibl. Bydd yr adroddiad yn cefnogi rheolwyr i ddarparu cyngor clir a chyson ar geisiadau, sy'n seiliedig ar dystiolaeth, wrth ddiogelu asedau gwymon.

Bwyd sy'n gyfrifol am y mwyafrif o weithgarwch cynaeafu â llaw, gyda swm cyfyngedig o wymon yn cael ei ddefnyddio i weithgynhyrchu colur a chynhyrchion fferyllol. Prin iawn oedd y dystiolaeth ar gyfer casglu gwymon a olchwyd i'r traeth yng Nghymru a Lloegr – gweithred y credir ei bod yn fanteisgar a thymhorol yn dilyn stormydd y gaeaf – i'w ddefnyddio fel gwrtaith neu gyflyrydd pridd. Mae ychydig bach o blanhigion aeddfed yn cael eu cynaeafu â llaw hefyd i ddarparu deunydd ffrwythlon ar gyfer tyfu gwymon. At ddibenion masnachol yng Nghymru a Lloegr, y rhywogaethau allweddol a dargedir yw gwymon lledr , (*Himanthalia elongata*) delysg (*Palmaria palmata*), rhywogaethau *Ulva*, rhywogaethau *Porphyra*, gwymon melys *(Chondrus crispus)*, *Fucus serratus* (ychydig o *F. vesiculosus*) a'r rhywogaethau morwiail *Laminaria digitata* a *Saccharina latissima*.

Mae gweithgarwch cynaeafu hamdden yn targedu ystod o rywogaethau, a cheir ychydig o orgyffwrdd â rhywogaethau a gynaeafir yn fasnachol. O blith y rhain, mae lafwr (rhywogaethau*Porphyra*) yn arbennig o bwysig i gynaeafwyr hamdden yn Ne Cymru, wedyn *F. vesiculosus*, delysg pupraidd (rhywogaethau*Osmundea*), morwiail, gwymon melys (*C. crispus* a *Mastocarpus stellatus* a *P. palmata*).

Mae gwybodaeth am gyfraddau twf, hanes bywyd, tymoroldeb ac atgenhedlu yn sail i'r broses adfer yn sgil cynaeafu. Mae'r rhain yn benodol i'r rhywogaeth ac i'r safle, a gallant amrywio dros amser. Nid yw gallu egin blanhigion i wasgaru wedi'i ddeall yn dda, ond mae'n fecanwaith adfer allweddol, a chredir ei fod yn isel yn achos y mwyafrif o rywogaethau. Ar gyfer rhywogaethau allweddol a dargedir, cyflwynir tystiolaeth o ddosbarthiad a'r paramedrau hanes bywyd hyn yn yr atodiadau, a hynny ar ffurf cyfres o goflenni rhywogaethau. Mae'r adroddiad yn nodi sut y dylai rheolwyr ystyried y rhain ac yn tynnu sylw at y gwahaniaethau rhwng rhywogaethau. Y gwymon codog bras (*Ascophyllum nodosum*) sy'n tyfu arafaf ac sydd fwyaf hirhoedlog o blith y rhywogaethau a gynaeafir, wedyn y gwymon morwiail *Laminaria hyperborea*, rhywogaethau eraill o forwiail a gwymon codog bras, a gwymon coch lluosflwydd. I'r gwrthwyneb, mae rhywogaethau *Ulva* a rhywogaethau *Porphyra*yn tyfu ac yn aeddfedu'n gyflym, i a gallant gytrefu creigiau sydd newydd eu clirio yn gyflym.

Mae gwymon yn cefnogi'r ecosystem naturiol a phobl o ganlyniad i brosesau cynhyrchu sylfaenol / cloi carbon, a chylchu maetholion sy'n sylfaenol i weoedd bwyd morol. Mae gwymon yn darparu ac yn addasu cynefinoedd ar gyfer ystod eang o organebau, gan gynnwys rhywogaethau o bysgod a physgod cregyn a dargedir yn fasnachol. Mae'r cyfraniad at weithrediad a gwasanaethau ecosystem yn amrywio, ac mae morwiail a rhywogaethau o wymon brown o bwysigrwydd ecolegol arbennig.

Mae effaith cynaeafu yn amrywio yn ôl y rhywogaeth, y raddfa, y dechneg a'r amodau amgylcheddol lleol. O ganlyniad i dynnu gwymon, mae cynaeafu yn lleihau cyfraddau twf a dynameg poblogaethau, ac mae hyn yn lleihau cylchu maetholion, yn lleihau argaeledd cynefinoedd, ac yn lleihau'r cynefinoedd a ddarperir. Er y gall adferiad ddigwydd mewn rhai achosion, gall newidiadau yng nghyfansoddiad y gymuned ddigwydd o ganlyniad i bwysau cystadleuaeth a phori. Dylid ystyried yr effeithiau hyn o fewn cynlluniau rheoli cynaeafu i liniaru effeithiau andwyol.

Mae rhywogaethau estron goresgynnol yn peri risg allweddol i rywogaethau a chynefinoedd brodorol. Nododd asesiad cyflym o'r dystiolaeth fod 33 o rywogaethau goresgynnol yn destun pryder, y maent yn debygol o fod yn gysylltiedig â gwymon a gynaeafir. Mae'r rhain yn cynnwys 13 o fathau o wymon goresgynnol, 13 o rywogaethau sy’n glynu neu’n clymu, a saith rhywogaeth symudol a allai gysgodi ymysg gwymon. Cafodd y risgiau a berir gan y rhain eu blaenoriaethu ar sail yr effaith ar rywogaethau a chynefinoedd brodorol, a pherthnasedd cynaeafu gwymon i wasgariad a’r dosbarthiad cyfredol. Nodwyd pedair ar ddeg o rywogaethau â blaenoriaeth, y gallai tair ohonynt fod o ddiddordeb masnachol. Mae mesurau i leihau'r risg o ledaenu yn cynnwys osgoi neu leihau sgil-ddalfa, ac awgrymir defnyddio offer ‘edrych, golchi, sychu’.

Mae’r dogfennau canllaw presennol ar gynaeafu yn seiliedig ar godau ymddygiad nad ydynt yn gyfreithiol rwymol. Bydd rheolaeth effeithiol yn benodol i’r safle ac i’r rhywogaeth. Gall dulliau rheoli fod yn wirfoddol (e.e. codau ymddygiad) neu'n statudol (e.e. cwotâu), a gellir eu crynhoi fel a ganlyn.

- Dulliau cynaeafu
  - Uchder torri, dylid gadael cyfran o'r planhigyn (cludafael a rhywfaint o ffrond) yn weddill ar y gwaelod
  - Torri'n ddetholus gyda siswrn yn hytrach na thynnu neu ddadwreiddio er mwyn hybu adferiad a lleihau sgil-ddalfa
  - Osgoi sgil-ddalfa o epiffytau a rhywogaethau sy'n agored i niwed
  - Osgoi cynaeafu deunydd atgenhedlol os yn bosibl (neu dim ond cymryd hanner o bob planhigyn yn achos H. elongata)
  - Yn achos rhai rhywogaethau (e.e. *for F. serratus* a *F. vesiculosus*), dylid cynaeafu rhan o blanhigion aeddfed yn unig
- Cyfnod cynaeafu
  - Cynaeafu yn ystod y tymor tyfu gweithredol
  - Osgoi cynaeafu yn ystod y tymor atgenhedlu
- Amledd cynaeafu
  - Cyfnodau braenar ar gyfer adfer y canopi yn achos *A. nodosum*a morwiail lluosflwydd
- Terfynau cynaeafu
  - Cwotâu, symiau neu derfynau bagiau
  - Cyfran y biomas stoc sefydlog a dynnwyd / adawyd yn weddill
- Ystyriaethau cynaeafu gofodol
  - Cynaeafu yn ofalus, gan adael planhigion nas cynaeafwyd rhwng y rhai a dynnwyd;
  - Siâp, maint a bylchau ardaloedd a gynaeafir (h.y. rhwng planhigion neu glytiau a gynaeafwyd);

Mae gwybodaeth am yr adnodd sydd ar gael yn hanfodol i’r dull o reoli cynaeafu gwymon â llaw yn gynaliadwy. Trafodwyd dulliau syml o asesu a monitro biomas a darparwyd ychydig o amcangyfrifon cychwynnol o fiomas gwymon rhynglanwol. Oherwydd y ceir bylchau yn y dystiolaeth, mae ansicrwydd yn parhau o ran cylch bywyd rhywogaethau penodol a'u gallu i adfer, yn enwedig gwymon coch a’ biomas stoc sefydlog yr holl rywogaethau sydd ar gael i'w cynaeafu. Mae'r adroddiad hwn wedi nodi dulliau rheoli sydd wedi’u cefnogi’n dda ac sy'n seiliedig ar ystyriaethau ecolegol, megis hanes bywyd a mecanweithiau adfer. Fodd bynnag, cydnabyddir y bydd angen i reolwyr fod yn rhagofalus o ran eu cyngor a’u gweithrediadau yn achos y mathau o wymon hynny lle ceir ansicrwydd ynghylch biomas a'r graddau y gellir eu hadfer.

Bydd rheolaeth effeithiol yn benodol i’r safle ac i’r rhywogaeth, gan ddefnyddio cyfuniad o ddulliau rheoli. Mae gweithgareddau cynaeafu gwymon yn rhoi cyfle i gynnwys rhanddeiliaid er mwyn cynnal prosiectau ymchwil cymhwysol, monitro safleoedd cyn cynaeafu, yn ystod cynaeafu ac ar ei ôl, ac i dreialu materion rheoli effeithiol. Gellid pennu bod gwaith monitro parhaus gan gynaeafwyr, ar ryw ffurf, yn un o amodau unrhyw drwydded ar gyfer manteisio ar stociau gwyllt.

Executive summary

Natural Resources Wales commissioned this literature review to improve understanding of the range of potential impacts of hand gathering seaweed and to review potential management measures. The report will support managers to provide clear, evidence based and consistent advice to applications while protecting the seaweed resource.

The majority of hand harvesting activity is for food use, with limited amount taken for cosmetics and pharmaceuticals. Little evidence for collection of beach cast weed was found in Wales and England, which is thought to be opportunistic and seasonal following winter storms, with applications as fertilizer or soil conditioner. A small amount of mature adult plants are also hand harvested to provide fertile material for seaweed cultivation. Commercially in Wales and England, *Sea spaghetti* (*Himanthalia elongata*), dulse (*Palmaria palmata*), *Ulva* spp., *Porphyra* spp., carrageen (*Chondrus crispus*), *Fucus serratus* (some *F. vesiculosus*) and the kelps *Laminaria digitata* and *Saccharina latissima* are the key species targeted.

Recreational harvesting activity targets a range of species, with some overlaps with commercially harvested species. Of these, Laver, (*Porphyra* spp). is of particular importance to recreational harvesters in South Wales, followed by *F. vesiculosus*, pepper dulse (*Osmundea* spp), kelps, carrageen (*C. crispus* and *Mastocarpus stellatus*) and *P. palmata*).

Growth rates, life history, seasonality and reproduction underpin recovery from harvesting. These are species and site specific and may vary over time. Dispersal potential of propagules is poorly understood, but is a key recovery mechanism and is thought to be low for most species. For key targeted species, evidence for distribution and these life history parameters is presented in the appendices as a series of species dossiers. The report identifies how management should take these into consideration and highlights differences between species. The wrack *Ascophyllum nodosum* is the most slow growing and long lived of harvested species, followed by the kelp *Laminaria hyperborea*, other kelp and wrack species, and perennial red seaweeds. Conversely *Ulva* spp. and *Porphyra* spp. are fast growing and quick to mature and able to rapidly colonise newly cleared rock.

Seaweeds support the natural ecosystem and people through primary production/ carbon fixation and nutrient cycling that underpins marine food webs. Seaweeds provide and modify habitats for a wide range of organisms, including commercially targeted fish and shellfish species. The contribution to ecosystem function and services varies with kelps and brown seaweeds being of particular ecological importance.

The impact of harvesting varies according to species, scale, technique and local environmental conditions. By removing seaweeds, harvesting reduces growth rates and population dynamics, this reduces nutrient cycling and reduces habitat availability and provision. While recovery may take place in some instances, changes in community composition can occur as a result of competition and grazing pressure. These impacts should be considered within harvesting management plans to mitigate adverse effects.

Invasive non-native species are a key risk for native species and habitats. A rapid evidence assessment identified 33 invasive species of concern, likely to be associated with harvested seaweeds, these include thirteen invasive seaweeds, thirteen attached or

fouling species and seven mobile species that may shelter amongst seaweeds. The risks from these were prioritised based on impact on native species and habitats, relevance of seaweed harvesting to dispersal and current distribution. Fourteen priority species were identified, three of which may be of commercial interest. Measures to reduce risk of spread include avoiding or reducing by-catch and 'check clean dry' equipment are suggested.

Existing harvesting guidance documents are centred around codes of conduct which are not legally binding. Effective management will be site and species specific. Management approaches may be voluntary (e.g. codes of conduct) or statutory (e.g. quotas), and can be summarised as follows.

- Harvesting methods
  - Cutting height, leave a proportion of the plant (holdfast and some frond) remaining at the base;
  - Selectively cut with scissors rather than plucking or uprooting to support recovery and reduce by-catch;
  - Avoid by-catch of epiphytes and vulnerable species
  - Avoid harvesting reproductive material if possible (or only take half from each plant in the case of H. elongata);
  - For certain species (e.g. *for F. serratus* and *F. vesiculosus*), only harvest part of mature plants
- Harvesting period
  - Harvest during active growing season;
  - Avoid harvesting during the reproductive season;
- Harvesting frequency
  - Fallow periods for recovery of canopy in *A. nodosum* and perennial kelps
- Harvest limits
  - Quotas, volumes or bag limits;
  - Proportion of standing stock biomass removed/left remaining
- Harvesting spatial considerations
  - Harvest sparsely, leaving unharvested plants between those taken;
  - Shape, size and spacing of harvested areas (i.e. between harvested plants or patches);

Knowledge of the available resource is essential for sustainable management of hand harvesting of seaweeds. Simple methods of assessing and monitoring biomass were discussed and some initial estimates of biomass of intertidal seaweed provided. Due to gaps in evidence uncertainty remains regarding the lifecycle and recovery capacity of certain species, particularly red seaweeds and the standing stock biomass of all species available for harvest. This report has identified management approaches that are well-supported and based on ecological considerations such as life-history and recovery mechanisms. However, it is recognised that advice and action by managers for those seaweeds where uncertainty exists regarding biomass and recoverability will need to be precautionary.

Effective management will be site and species specific, utilising a combination of management approaches. Seaweed harvesting activities provide an opportunity to involve stakeholders to conduct applied research projects, monitoring sites before, during and after harvesting and to trial effective management issues. Continued monitoring in some form by harvesters could be a condition of any license to exploit wild stocks.

1. Introduction

Seaweed harvesting is a growth area, with uses in food, textiles, pharmaceuticals and as organic fertilisers. Seaweed is also in increasing demand for cosmetics and the potential for its use in biofuel is also expanding. The number of businesses operating commercial harvesting activities is rapidly evolving, and was last comprehensively reviewed for the UK by Capuzzo & McKie (2016). Seaweed industry organisations now exist in the UK including Seaweed Forum Wales, Scottish Seaweed Industry Association, and the Irish Seaweed Consultancy.

Around 650 species of seaweed have been recorded in the UK. Seaweeds in Wales and England can broadly be grouped into the following categories: kelps (family Laminariaceae); brown seaweeds; red seaweeds; green seaweeds; and calcified seaweeds (e.g. maerl). This report focuses only on the species which are targeted for hand-harvesting, both commercially and recreationally (taken for personal use and not for sale). The majority of these are collected fresh from the shore, but a mixture of species may also be gathered from strand lines as beach cast or drift weed.

1.1 Background

Natural Resources Wales (NRW) and Natural England currently advise on a range of hand gathering enquiries / applications of various sizes. Currently advice is given by NRW and Natural England specialists on the specific enquiry / application using the best available evidence and knowledge, using the precautionary principle.

1.2 Aims

NRW commissioned this report to improve their understanding of the range of potential impacts of hand gathering seaweed, in order for staff to be able to provide clear, evidence based and consistent advice to applications while protecting the seaweed resource.

Specifically, the aim was to increase NRW and Natural England’s understanding of the ecology of potentially harvested seaweeds and potential impacts of hand gathering, and to provide an evidence based review of potential management measures to inform decision making regarding appropriate and sustainable levels of collection. It is anticipated that this report will provide both organisations with the evidence base required for staff to confidently and consistently respond to casework.

1.3 Outputs

This project was a desk-based contract to provide a thorough evidence-based examination of various components of the ecology and management of seaweeds to inform decisions on sustainable harvesting. The contract considered a range of information sources, focussing mainly on published scientific literature and grey literature. The project outputs consist of this report.

1.4 Report structure

This report consists of an introductory section and methodology, and three principal evidence sections: Section 2 provides a review of key harvested species and relevant biological and ecological factors that should be considered in management plans; Section 3 presents a review of hand harvesting methods, guidance and ecological implications of harvesting and Section 4 outlines management approaches. The report concludes with final summary sections on knowledge gaps, recommendations (Section 5) and conclusions (Section 6).

Detailed technical information is presented in the report appendices (1-14). These include a review for each of the principal harvested species of their biology, ecological importance, implications of harvesting and any associated management and guidance to assist decision making on sustainable harvesting limits. These species dossiers are intended to be a key resource for managers to support the development of species specific management plans.

2. Key harvested seaweed species, distribution, life history and ecological importance

This report has focussed on key targeted species and detailed evidence for these is presented in Appendices 1-14. These are *Alaria esculenta, Laminaria digitata, Laminaria hyperborea, Saccharina latissima, Saccorhiza polyschides, Ascophyllum nodosum, Fucus serratus, Fucus vesiculosus, Himanthalia elongata, Chondrus crispus, Mastocarpus stellatus, Palmaria palmata, Porphyra spp. and Ulva spp.*

Additional species of lesser commercial importance, for which limited evidence of harvesting was found, were not reviewed. These include *Pelvetia canaliculata*, *Corallina* spp., *Fucus spiralis*, *Gracilaria* spp., and *Osmundea* spp (see Table 1). Sensitivity reviews, which contain life history information and recovery rates for these species are available on the MarLIN website.

2.1 Species harvested in Wales and England

Commercial operators are likely to target selected, high demand species with up to 20 species known to be harvested commercially in Wales and England (see Table 1 and Morris-Webb in prep). Commercially in Wales and England, the kelps *Laminaria digitata* and *S. latissima*, the brown seaweeds *H. elongata*, *F. serratus* (some *F. vesiculosus*), the red seaweeds, *P. palmata*, *Porphyra* spp., *Chondrus crispus* and the green seaweed, *Ulva* spp. are the key species targeted.

Collection of living resources is difficult to assess as it is rarely the focus of research or monitoring. In Wales, surveys of intertidal harvesters in 2018-19 found that a total of 23 different algal taxa were targeted (Morris-Webb in prep). Of these, harvesters assessed *Porphyra* spp., *F. vesiculosus*, *Osmundea* spp., kelps, *C. crispus* and *P. palmata* to be of particular importance. *Corallina* spp. and *S. polyschides* are potentially collected, particularly recreationally, but are not currently known to be commercially targeted. The motivations to collect species were generally for food, cosmetics, or monetary value but also for uses such as aquaria and fertiliser (Morris-Webb in prep).

Although they are not currently commercially targeted, it may be possible that, as their ranges extend further through the UK, the non-native kelp *Undaria pinnatifida* (Wakame), which is popular in Asian cuisine, and the warm water Golden kelp *L. ochroleuca* could be targeted in future harvesting activity. While the sale of harvested *U. pinnatifida* is currently not permitted under Schedule 9 of the Wildlife and Countryside Act, it is already harvested in parts of England for personal use.

The sources for common names in Table 1 are the MarLIN website, supplemented by the project team. *S. polyschides* is not taxonomically speaking a true kelp, but is ecologically and functionally similar and is therefore included as ‘kelp’, throughout the report.

Table 1. Seaweed species that are harvested in Wales and England, showing scientific and common names. For species with detailed evidence reviews the relevant appendix number is provided.

Scientific name	**Common name**	**Appendix No**
Alaria esculenta	Atlantic Wakame, Dabberlocks, Badderlocks, Winged kelp	1
Ascophyllum nodosum	Knotted wrack, Egg wrack, Rock weed	2
Chondrus crispus	Irish moss, Carrageen moss	3
Fucus serratus	Toothed wrack, Serrated wrack	4
Fucus vesiculosus	Bladder wrack, Black tang, Rockweed, Bladder *Fucus*	5
Himanthalia elongata	Thongweed, Sea thong, Sea spaghetti, Sea bean.	6
Laminaria digitata	Kombu, Oarweed	7
Laminaria hyperborea	Kombu, Tangle, Cuvie, Forest kelp	8
Mastocarpus stellatus	False Irish moss, Carrageen	9
Palmaria palmata	Dulse	10
Porphyra spp.	Laver, Nori, Laver bread “Bara Lawr”	11
Saccharina latissima	Sugar kelp, Sea-belt, Devil’s apron	12
Saccorhiza polyschides	Furbellow	13
Ulva spp.	Genus includes; Sea lettuce, Gut weed	14
Corallina species	Coral weed	No appendix
Gracilaria spp. (including *Gracilaria vermiculophylla)*		No appendix
Fucus spiralis	Spiral wrack, Flat wrack	No appendix
Osmundea species	Pepper dulse	No appendix
Pelvetia canaliculata	Channelled wrack	No appendix
Sargassum muticum	Wire weed	No appendix

2.2 Geographical distribution and habitat preferences of key harvested seaweed species

The geographic distributions of each seaweed species is detailed in the species appendices (Appendices 1-14). Broadly, the majority of seaweeds form an attachment to hard substratum such as rocks or boulders, so are likely to be absent from large areas of south eastern England which are typically dominated by soft-sediment habitats. The section on climate change below outlies some species that are close to range edges in the UK and the species dossiers (Appendices 1-14) provide further species distributions and UK maps.

In summary, species with a northern distribution, with southern limits in the southern UK and into France, include the kelps, *A. esculenta*, *L. digitata*. Species that are relatively central within their range in Wales and England, are, *A. nodosum*, *C. crispus, F. serratus*, *F. vesiculosus*, *H. elongata, L., hyperborea*, *M. stellatus, P. palmata*, *Porphyra* spp. *S. latissima*, S. *polyschides*, *Ulva* spp.

High wave exposure on open coasts in Wales and south western England generally favours the brown seaweeds, *F. vesiculosus* and *F. serratus* with reds *C. crispus, Mastocarpus stellatus* and *P. palmata* in midshore intertidal habitats. Unusually for seaweeds, *Porphyra* spp. appears to benefit from sand scour and emersion, either through enhanced growth or reproduction, or by reduced competition or predation (Littler *et al*., 1983; Airoldi, 2003). In low intertidal and shallow subtidal habitats, *A. esculenta* is found only at the most wave exposed sites, with *L. hyperborea* and *L. digitata* thriving in moderate-to-high energy conditions. As shelter from wave action increases, *Ascophyllum nodosum* and *Saccharina latissima* become dominant, while *F. vesiculosus* may outcompete *A. nodosum* on very sheltered shores.

Larger-scale geographical distributions have implications for harvest management, as species at the range edge are likely to experience a range of environmental stresses which may reduce their growth rate and recovery following harvest. For example, increasing sea temperature was found to have the largest negative impact on yields of *L. hyperborea* in Brittany, where the species is close to its southernmost range edge (Werner & Kraan 2004; Davoult *et al*., 2011).

Climate change and future range shifts

Warming seas can result in changes in distribution which have been widely documented for a variety of seaweed species (Cosson, 1999; Husa, 2007; Smale, 2020; Fernandez, 2011; Teagle & Smale 2018; Pessarrodona *et al*., 2019), resulting in declining yields or local extinctions. Most notably in the UK, the warm-adapted *L. ochroleuca* has shown a northward range expansion and was recently recorded in Ireland for the first time (Schoenrock *et al*., 2019). Some cold-water species, including *L. digitata* and *A. esculenta*, are found in the UK (i.e. southern England) towards their southernmost range edges and, as such, are close to their upper thermal limit. Marginal populations of *L. digitata* are declining in France (Raybaud *et al*., 2013) and productivity and growth rates are diminished in southern England, compared to range-centre populations (King *et al*., 2020). Similarly, anecdotal evidence suggests that the spatial extent and abundance of *A. esculenta* has declined towards its southern range edge (Hiscock *et al*., 2004), leading to small, fragmented populations.

With warming conditions and increased storm disturbance to seaweed canopies, it is anticipated that fast growing *S. polyschides* will increase in abundance, as it has in south England and Brittany, potentially increasing competitive pressure on *L. digitata* and *L. hyperborea* at their southernmost range (Smale *et al*., 2013). Gaps created by mechanical harvesting can be colonised by *S. polyschides* (Werner & Kraan 2004; Mac Monagail *et al*., 2017). *S. polyschides* can tolerate higher temperatures than *L. digitata*, and even a slight increase may positively affect growth and reproduction of *S. polyschides*, and negatively impact *L. digitata* (Werner & Kraan 2004).

Seaweed life history

Seaweed life histories vary, with harvested seaweed species ranging from long-lived, slow growing perennial species to annual species, that grow rapidly over a season. *A nodosum* is a long lived, perennial species. The holdfast has been demographically modelled to last centuries (Åberg, 1992), with individual fronds lasting for approximately 10-20 years (Stengel & Dring, 1997), and clumps of plants reported being 60 - 550 years old (Seip 1980; Åberg 1992). *A. nodosum* takes five years to become sexually mature (Sundene 1973). The kelp *L. hyperborea* is also long lived, reaching 11-20 years old, and slow to mature at 2-6 years. Relative to Laminarial kelps, *A. esculenta* is fast growing and fairly short lived

By contrast, opportunistic *Ulva* spp. can reach maturity in a matter of weeks, which along with its high reproductive output and dispersal capacity allow it to rapidly recover or colonise new areas. These differing life-histories have clear implications for recovery following harvesting.

Other brown and red seaweed species tend to be intermediate being perennial but shorter lived than kelps and reaching sexual maturity sooner. Key species life history parameters are summarised in Table 2, detailed species information is provided in appendices 1-14.

Reproduction and dispersal

Species specific information on reproduction for key harvested species is provided in Appendices 1-14. Seaweeds may reproduce sexually and asexually through vegetative propagation, by producing new shoots from the base of the parent plant. Vegetative propagation is more common than sexual reproduction in *A. nodosum,* with vegetative growth rather than sexual reproduction being key for maintenance of populations, and early mortality exceeding 99.9% (Aberg, 1996; Dudgeon & Petraitis, 2005). *C. crispus* spores also have low viability (<30%), suggesting that populations are maintained by vegetative growth (Scrosati *et al.,* 1994).

All kelps have a gametophyte life stage, which are microscopic, filamentous forms that generally live attached to rocky substrates beneath the kelp canopy. Although this stage typically lasts for only ten days (Norton 1992), if conditions for germination are not optimal gametophytes can grow vegetatively, potentially for several years. This creates a microscopic seed bank from which visible plants can develop, eventually appearing as small sub canopy juvenile plants.

A. nodosum can reproduce sexually or by vegetative propagation, where a new individual develops directly from the basal shoots of the parent plant. vegetative propagation is the most common and important method of reproduction for this species (Aberg 1996).

The dispersal of seaweed spores is of vital importance to post-disturbance recruitment and recovery. The dispersal capability of seaweed spores will depend on the species, release depth (based on where the fruiting bodies are on the plant), season, and local hydrodynamic conditions.

Relatively speaking, all seaweeds produce a large numbers of spores. The dispersal potential of seaweeds is poorly understood, but is generally thought to be low for many species. The zygotes of *H. elongata* are very large in comparison to most seaweeds, and those of *F. serratus* are negatively buoyant, so they rapidly settle onto the seabed, limiting dispersal. Spores of the red seaweeds such as *C. crispus* settle within 10 m of the parent plant (Scrosati *et al.,* 1994) and dispersal of Fucoids is effective over short distances (Arrontes 2002).

Kelp zoospores (motile spores) remain in the water column for up to 24 hours (Werner & Kraan 2004) and can reportedly travel 10s to 100s of meters from the parent plant, but must settle in very high densities for successful fertilisation. By contrast, *Ulva* spp. spores, referred to as “swarmers”, can successfully recruit over distances of at least 35 m (Amsler & Searles, 1980).

An alternative dispersal mechanism is the spread of mature adults which become dislodged and float on water currents as rafting drift plants, releasing spores. Establishment of new populations requires high spore settlement densities, which could potentially occur where eddies retain drift plants within an area. This mechanism has been suggested for the spread of certain species, such as the non-native kelp, *U. pinnatifida*. This method is more likely to be effective for species such as *F. vesiculosus* which has air bladders allow them to readily float, whereas rafting is unlikely in negatively buoyant species, for example *F. serratus* (Coyer *et al*., 2006).

All species have some microscopic life stages, potentially allowing for recovery from an invisible “seed bank”.

Recovery potential

Recovery from harvesting will be influenced by the proportion of the plant removed, harvesting intensity, frequency and proportion of standing stock biomass harvested. Growth, reproduction and recovery rates of harvested species are detailed in Appendix 15 and the species dossiers (Appendices 1-14) provide more detailed information for key targeted species. Knowledge of the different growth and recovery rates between species is key to developing species specific management plans.

An overview of recovery rates is provided in Table 2 (below) with more detailed information presented in Appendix 15. The recovery categories are based on those used by the Marine Life Information Network (MarLIN) (see Appendix 15).

Most kelps are long lived (with *S. polyschides* and to a lesser extent *S. latissima* as exceptions). Provided that the holdfast, stipe, meristematic tissue at the base of the frond remains, the frond can regrow. They exhibit high growth rates and productivity, and maintain biomass despite natural processes of erosion and frond loss from the tips.

A. nodosum and *Fucus* spp. both grow from the apical tip (the end of the frond). *A. nodosum* is particularly slow growing and long lived, forming only one bladder per year, although can also regenerate from the base, provided that sufficient frond length remains. Mac Monagail *et al*. (2017) report that cutting at a certain height appears to stimulate dense, bushy growth, in *A. nodosum,* similar to pruning of terrestrial crops. The wrack *F. vesiculosus* and the red seaweed *P. palmata* both demonstrate prolific growth from cut or damaged frond tips or edges, as well as regenerating from the perennial holdfast.

Opportunistic species such as *Porphyra* spp. and *Ulva* spp. are fast growing, and likely to recover quickly during the growing season (spring and spring-summer respectively), provided the holdfast and base of the frond remains.

Hand harvesting codes of conduct and guidance generally advise cutting at a specific height, leaving the basal parts to regenerate (see Section 4.3 and Appendix 16), which is likely to result in much more rapid recovery than removal of whole individuals. If the entire plant is removed, for example by plucking or uprooting, from large patches of the shore, populations of most perennial seaweeds (long lived reds, fucoids, kelps) may recover by either plant growth directly from the microscopic ‘seed bank’ life stages or by settlement and recolonization from nearby source populations, which is likely to be slow (see recruitment section below). If, however, plants are only patchily removed from the canopy, recovery may be relatively rapid. In some kelp and wrack species (for example *L. hyperborea, H. elongata,* and *F. serratus*) removal of the adult canopy allows more light into the understory, which in pristine habitats allows for rapid growth of juveniles or germlings (d'Avack & Garrard, 2015; Stagnol *et al.* 2015; 2016)

Long lived, large, slow growing species such as kelps (particularly *L. hyperborea*) and *A. nodosum* are considered as climax or late succession species. They are generally slower to recolonise new areas following removal, for example after clearance of *A. nodosum,* the species had not returned to dominance within 12 years, with a mixed assemblage of *Fucus* spp. and *A. nodosum* present (Sundene 1973; Choi & Norton 2005). *Fucus* spp., *M. stellatus* and *C. crispus* are mid-successional species, while *P. palmata* is relatively fast growing and opportunistic, benefiting from intermittent disturbance. By contrast to climax species, opportunistic *Ulva* spp. can grow rapidly, particularly in summer, quickly recolonising totally cleared areas substantial distances from source population.

Recovery rates of the algal canopy are summarised in Table 2. The recovery categories are defined in Appendix 15. Recovery rates are indicative and based on evidence where available supplemented by information on life-history traits (e.g. recolonisation and growth rates) where necessary. They are based on an assumption that it is a single harvesting event with best practice methods followed (see Appendix 16 for suggested management guidance); recovery from total clearance is based on small (2m^2) areas with adult source populations nearby to supply propagules. Recovery rates following more intensive, repeated or extensive harvesting, for example where large areas of shore are denuded of seaweed cover, will be slower than those presented in the table. The evidence column indicates the level of confidence based on the available literature.

Some key mechanisms and evidence that underpin recovery are outlined briefly below. Vegetative growth is likely to be an important recovery mechanism for some species such as *F. vesiculosus* (McLachlan & Chen, 1972). *P. palmata, Ulva* spp. and *Porphyra* spp. that can all regrow from small fragments.

The red seaweeds *P. palmata, C. crispus* and *M. stellatus* all have a basal crustose phase which appears as a flat, dark red-black crust covering rocks, with a rubbery appearance, which was once thought to be a distinct species (Bunker *et al.,* 2010). The bases have greater longevity than the frond, and are capable of regenerating new fronds after damage or loss.

Early mortality can be high in some species, for example survival chances of *A. nodosum* beyond the first year is less than 0.01%, with recovery more likely by vegetative growth from the base of existing adults (Aberg 1996; Dudgeon & Petraitis, 2005). Survival of *C. crispus* spores is also low (<30%) (Scrosati *et al.,* 1994), whereas *Fucus* recruitment over short distances is very efficient.

Table 2. Recovery rates of harvested species, simplified from Appendix 15. Sources: MarLIN (ww.marlin.ac.uk); Burrows *et al*., 2018. Recovery from partial harvest assumes best practice guidance is followed, Please note, recovery from total clearance refers to small patches only.

Scientific name	**Growth habit**	**Growth season**	**Lifespan (years)**	**Recovery (partial harvest)**	**Recovery (total clearance in small 2m^2 patches)**	**Evidence**
A. esculenta	Perennial	Spring	4-7 years	High	High	Low
L. digitata	Perennial	Winter	4-6 years	High	High	High
L. hyperborea	Perennial	Winter	11-20 years	High	Medium	High
S. latissima	Annual / perennial	Winter	2-4 years	High	High	High
S. polyschides	Annual	Spring	8-18 months	High	High	Medium
A. nodosum	Perennial	Summer	>60 years	Medium	Low	High
H. elongata	Bi-annual	Spring	2-3 years	High	High	Low
F. serratus	Perennial	Spring	2-5 years	High	High	Medium
F. vesiculosus	Perennial	Summer	2-5 years	High	High	High
C. crispus	Perennial	Summer	2-6 years	High	Medium	High
M. stellatus	Perennial	Summer	Presumed 2-5 years	High	Medium	Low
P. palmata	Perennial	Summer	Unknown	High	High	Low
Porphyra spp.	Annual	Spring	Unknown	High	High	Low
Ulva spp	Annual	Summer	Unknown	High	High	Medium

In addition to regeneration from holdfasts and encrusting perennial life stages and production of spores for dispersal and recruitment, many species have microscopic life stages that serve as 'seed banks' from which germlings can appear following harvesting or disturbance.

Sub-canopy juveniles can play an important role in recovery of *L. hyperborea, H. elongata,* and *F. serratus* habitats, whereby removal of the adult canopy allows more light into the understory, which in pristine habitats allows for rapid growth of juveniles or germlings (d'Avack & Garrard 2015; Stangol *et al.* 2015; 2016).

While under canopy germlings and juveniles are clearly a valuable recovery mechanism for kelps it should not be considered an unlimited supply, as its existence is dependent on the presence of a healthy population of reproductively mature adult plants nearby.

The opportunistic red seaweed *Porphyra sp*. has a complex lifecycle which includes a microscopic sporophyte called the conchocelis stage, which until recently was thought to be a separate species. The conchocelis burrows into wood, rock or the shells of molluscs, growing vegetatively (Knoop *et al.,* 2020). This cryptic stage allows *Porphyra* to survive periods of intense disturbance, such as sand scour or grazing, which removes adult plants. Sudden blooms can form when conditions are suitable, such as following winter storms when bare rock becomes available (Knoop *et al.,* 2020; Robles, 1982), which may buffer populations from some level of harvesting pressure (Nelson & Conroy, 1989).

Recruitment and survival of young germlings can be promoted by the presence of adults in *A. nodosum, H. elongata* and *Fucus* spp., with *Fucus* propagules settling in high densities close to or beneath parent plants (Arrontes, 2002; Knight & Parke, 1950). The canopy provides protection from desiccation, high temperature and high irradiances, although shading may limit juvenile growth (Stengel, pers. Comm. in White 2008). The adult *A. nodosum* canopy may also benefit recruitment by excluding competing seaweed species such as *Fucus* spp. and Coralline algae. Indeed, hand harvesting of *A. nodosum* (near the base with a sickle) resulted in short-term colonisation by *F. vesiculosus* and *Ulva* spp. prior to full recovery of *A. nodosum* within 3-5 years (Tyler 1994). The presence of red algal turfs including *Corallina* spp. can then prevent recovery of fucoids and other species by inhibiting recruitment. Interestingly, recruitment of *F. serratus* in some instances occurs under a canopy of *Ulva* spp., which protected the young plants from wave action (Knight & Parke 1950), but in other cases, the presence of *Ulva* spp. inhibits the colonisation of *F. serratus* (Sousa 1979) suggesting tempo-spatial variation in interaction.

Grazing pressure also plays a key role in recovery and early mortality. Exclusion of grazing limpets on shores in southern Britain, led to the colonisation of red algal turfs by *H. elongata* and *F. serratus* within 2 years (Boaventura *et al*. 2002). Mrowicki *et al*. (2014) found that limpet and barnacle removal allowed ephemeral and fucoid seaweeds to establish on both sheltered and wave exposed shores in Ireland. Further, following the Torrey Canyon oil spill which killed intertidal grazers, *Ulva* spp. rapidly recruited, followed by *Porphyra* spp. two months after the spill (Smith 1968).

Season of recruitment will also influence recovery. For example, *S. polyschides* is generally considered opportunistic and readily able to colonise gaps in the canopy. However experimental clearances conducted in August, when no spores of the species were released, resulted in colonisation by red algae potentially inhibiting recruitment of *S. polyschides* (Kain, 1975).

Seasonal and regional influences on recovery rates

Most seaweed species demonstrate seasonal growth, so harvesting season will impact recovery, with faster recovery expected during times of peak growth. However, periods of peak growth, reproduction and recruitment vary greatly between species. Understanding these aspects is key to developing species specific management plans, detailed evidence for key targeted species is provided in Appendices 1-14. Some illustrative examples of differences between species are provided below with examples of appropriate harvesting periods.

The kelp, *L. hyperborea* exhibits highest growth rates in winter and spring, culminating in a shedding of a collar or old growth (termed the ‘May cast’). Following the May cast, growth rates are negligible for most of the rest of the year. Rates for *L. digitata* and *A. esculenta* peak later in the year and growth continues (albeit at reduced rates) for a longer period. There is a narrow window for harvesting *A. esculenta* in late spring, when plants have reached their maximum biomass but before the blade begins to erode away from June through the summer. In *S. polyschides*, a kelp which behaves as an annual, seasonal biomass variation is extreme, with only the holdfast remaining through winter.

Conversely, *L. ochroleuca* and *S. polyschides* exhibit greatest growth rates in late spring through to autumn, when light levels are highest (Burrows *et al*., 2018; Pessarrodona *et al*., 2021).

For *A. nodosum* growth is slowest in the winter (November and December), and fastest in the spring (April and May). Growth rates tend to be slowest at the extremes of the tidal range, meaning that they grow slower on the lower and upper shores, and fastest in the mid-shore (Stengel & Dring 1997). Growth of *A. nodosum* is slow, plants typically require 5-8 years to reach their full size (Seip 1980; Aberg 1992).

The harvestable ‘straps’ of *H. elongata* are entirely seasonal, growing rapidly between February to May. The straps are lost from October on exposed shores, but may persist until February at more sheltered sites (all be it in a condition that may not be desirable to harvest). Similarly to *L. digitata*, the brown seaweed *F. serratus* may be present on the shore at high densities (covering 95% of suitable habitat) during the summer, becoming fragmented and patchy as a result of dislodgement during winter storms. Growth rates of *F. vesiculosus* are also much higher in summer, and on sheltered shores are double that of exposed sites (Carlson 1991; Knight & Park 1950).

For the small perennial red *C. crispus*, growth occurs from May to November in Canada (Juanes & McLachlan, 1992; Chopin *et al*. 1999), although little seasonal variation in growth was found in populations in Ireland (Pybus 1977). In the USA, *C. crispus* harvested in summer was found to recover relatively quickly (5-6 months), while recovery following winter harvest was much slower (Mathieson & Burns 1975). These seaweeds reproduce in the UK between autumn to spring (Dickinson, 1963; Pybus, 1977), so a harvesting strategy focussed during the summer months is likely to not only be followed by faster recovery of biomass but could potentially have less of an impact on reproduction. *M. stellatus* is thought to follow a similar pattern to *C. crispus,* although limited evidence was found.

Populations of *P. palmata* from Spain also grow between March-August, with fronds breaking between August to March, although no information was found for growth of this species in the UK.

For more opportunistic species such as *Ulva* spp. and *Porphyra* spp., very rapid growth occurs during seasonal blooms. Outside of these periods the harvestable resource will be relatively limited.

In addition to seasonality of biomass or vegetative growth, reproduction peaks seasonally in many species (see Appendix 15 for individual species detail). For example, while reproduction can occur all year, Knoop *et al.* (2020) summarise that for *Porphyra dioica* in Wales*,* recruitment peaks in autumn and winter, followed by vegetative growth and reproduction in spring.

Norwegian studies of kelp recovery following mechanical harvesting have demonstrated high site-level variability, with populations recovering in ~4 years at one site and >6 years at another (Christie *et al.,* 1998). Growth rates of Norwegian *L. hyperborea* are higher in wave exposed locations (Sjøtun *et al*., 1998), therefore, harvested kelp can recover more rapidly in wave exposed locations than in sheltered locations. It is evident that local environmental conditions mediate recovery and promote site-level variability. Interestingly, the season in which *L. hyperborea* is harvested does not appear to have a strong effect on the rate of recovery, and similar patterns of species succession are observed for all seasons following experimental clearance (Kain 1975; Christie *et al*., 1998).

Inter-annual variation in seaweed productivity must also be taken into account for management, as during years with poor environmental conditions for growth the seaweed standing stock is likely to be less than anticipated. Contingency plans need to include allowances, for example in order the achieve a target biomass yield, collectors may harvest from a greater area, or the same area more intensely.

2.3 Ecological importance of key harvested seaweed species

Ecosystem function, services, goods and benefits classification

Key ecological roles of harvested seaweed species are described below. The species appendices provide evidence for each species. The sections below do not strictly follow any existing frameworks but development and review considered the ecosystem service frameworks developed by Potts *et al.,* (2014) and the Common International Classification of Ecosystem Services (CICES v5.1; Haines-Young & Potschin, 2018). In ecosystem service assessment frameworks, ecological processes are sometimes termed intermediate or supporting services whilst those that directly benefit humans are described as final ecosystem services or goods and benefits.

In general, long-lived canopy forming kelps and brown algae (i.e. *L. hyperborea*, *A. nodosum*), which represent the climax community of succession, provide a greater range of ecological functions, and to a greater extent due to their larger size than smaller and shorter lived red and green algae.

More evidence was available for kelps, in particular *L. hyperborea* (e.g. Smale *et al*., 2013) than other species. Ecosystem services and goods and benefits have not been consistently assessed across species and many aspects are unstudied for most species. Habitat provision appears to be the most frequently studied aspect, as it is readily examined and quantifiable.

Primary production and nutrient cycling

Kelp forests are acknowledged as one of the most productive ecosystems on earth (Dayton 1985, Steneck *et al.,* 2002, Smale *et al.,* 2013). In the Atlantic, kelp primary production (the assimilation and fixation of inorganic carbon and nutrients (carbon dioxide, water and mineral nutrients) into organic compounds, including algal structural biomass), can be in excess of 1,000 g C m-2 year-1 (Mann 1973, Smale *et al.,* 2013), and that from Laminaria species has been estimated at between 110 and 1,780g/Cm-2yr-1 (Mann 1973). Kelp beds may account for~45% of primary production in UK coastal waters, and 12% of marine production in the entire UK Exclusive Economic Zone (Smale *et al.,* 2013).

Limited evidence was found on the primary production rates of brown (fucoid) red and green algae. Brown algae, as large primary producers capable of fast growth, are likely to provide high levels of primary production (Kelly, 2005) while production by the smaller green and red algae is likely to be lower.

Primary production supports food chains, either directly where the algae are directly grazed, or indirectly through transport of organic matter in the form of detritus or dissolved organic matter. The role of seaweeds, particularly kelps in organic matter transport and the importance of this is becoming clearer. Herbivory is generally low in kelp forests, with less than 10% of live kelp biomass thought to be consumed directly by grazers (Norderhaug & Christie, 2011), and 80% being exported as detritus (Burrows *et al.,* 2014; Wernberg & Filbee-Dexter, 2018).

Seaweeds exude considerable amounts of dissolved organic carbon which are taken up readily by bacteria and may even be directly consumed by some larger invertebrates. Dissolved organic carbon, algal fragments and microbial film organisms are continually removed by the sea. This may enter the food chain of local, subtidal ecosystems, or be exported further offshore. Detritus is produced through incremental blade erosion, fragmentation of blades, and dislodgement of whole fronds and thalli. Detritus settles within kelp beds or forests and is exported to neighbouring or distant habitats, including sandy beaches, rocky intertidal shores, rocky and sedimentary subtidal areas, and the deep sea. Exported kelp detritus can provide a significant resource subsidy and enhance secondary production in these communities ranging from tens of meters to hundreds of kilometres from the source of production. Detritus or dissolved organic material transport can provide valuable nutrition to potentially low productivity habitats such as sandy beaches (Smale *et al.,* 2013).

Primary production rates vary between species, seasons and regions (Pessarrodona *et al.,* 2019). Deriving estimates of standing stock biomass, and therefore primary production, is challenging because the biomass density and the cross- shore width varies greatly with species, time (both seasonally and inter-annually) and location (both within and among sites) (Reed and Brzezinski, 2009).

Secondary production

Seaweeds make a direct contribution to the food of many marine species through grazing of primary production (largely the algae biomass) and the production of planktonic propagules which contribute to pelagic food chains.

Fish, otters, seals and diving seabirds and sea ducks feed within kelp forests due to the high biomass and biodiversity associated with kelp and the subsequent food availability (Kelly, 2005). In Norway, small gadoid fish are more abundant in unharvested *L. hyperborea* beds and cormorants preferentially foraged within kelp-forested areas and performed significantly more dives when feeding in harvested versus un-harvested areas suggesting lower foraging yield (Lorentsen *et al*., 2010). The shallow fringes of kelp forests are also an important area for feeding birds, in particular wading birds (Kelly, 2005).

Carbon sequestration

Primary production can be transported as organic and inorganic carbon to deeper ocean waters. This process is referred to as the ‘biological carbon pump’. Marine primary producers contribute at least 50% of the world’s carbon fixation and may account for as much as 71% of all carbon storage (Chung *et al.,* 2011). Seaweeds, due to their high rates of production, fragmentation, and ability to be transported, appear to be able to make a significant contribution to carbon storage and sequestration (Trevathan-Tackett *et al.,* 2015; Abdullah *et al.,* 2017). However, the fraction of exported seaweed carbon that is actually stored on meaningful timescales is poorly resolved and is likely to be highly context-specific, depending on the seaweed species, rates of carbon release, hydrodynamics and proximity to storage habitats. Evidence from a handful of studies suggests that 4-16% of seaweed carbon may be sequestered (Krause-Jensen & Duarte, 2016).

Habitat provision

The degree of habitat provision and biodiversity associated with harvested species varies according to size and complexity and the longevity of the species as well as the environment. Shape and structural complexity of macroalgae are important factors in determining patterns of abundance and size structure of associated epifaunal organisms.

The most structurally complex algae harbour more abundant and diverse assemblages of invertebrates because among other effects, they provide a larger availability of surface for colonisation by fauna and epiphytic algae (Cacabelos *et al.,* 2010 and references therein). The more subtidal kelps and their structural complexity providing holdfasts, stipes and blades supports the highest diversity of understorey, holdfast community and epifauna and flora (King *et al.,* 2020). Intertidal species, particularly those that are smaller and less structurally complex support less species. For example, *Porphyra* spp. which forms less complex habitats is associated with fewer species than the more upright red seaweeds *M. stellatus* and *C. crispus* (Hacker & Steneck. 1990), which in turn are associated with fewer species than fucoids. The correlation between associated species can be directly attributed to the size and structure of the algae. The number of species associated with species can also vary over environmental gradients such as the changes in diversity associated with *S. latissima* biotopes within the UK Marine Habitat classification (Connor *et al*., 2004) according to degree of sedimentation.

Long-lived habitat forming species are intuitively likely to have increased species richness than those provided by shorter lived species, although this can be offset by complexity of microhabitats associated with the seaweed which may increase diversity. For example, the annual kelp *S. polyschides* has been found to support lower levels of biodiversity than perennial Laminarian kelps (McKenzie & Moore, 1981). However, the unusual bulbous holdfast of *S. polyschides* creates a high volume micro-habitat, a suitable refuge for larger organisms (i.e. predatory fish, crustaceans) that are typically absent from *L. hyperborea* (McKenzie & Moore, 1981).

By providing a greater availability of niches and increasing the number of microhabitats, habitat forming species offer refuge from environmental and biological stress (Hixon & Menge 1991; Hauser *et al.,* 2006; Almany, 2004; McAbendroth *et al*., 2005). A greater level of microhabitats with increased resources and shelter can alter predation and competition interactions within an assemblage by reducing conflictions, allowing species to coexist (Hixon & Menge 1991). Smaller perennial species such as fucoids and long lived red seaweeds in the intertidal provide shade and shelter from predation and desiccation. Recent surveys in UK kelp forests have recorded high diversity of faunal and floral assemblages, most likely due to biogenic habitat provision and modification by foundation kelp species (Teagle *et al*., 2018, Bué *et al*., 2020, Smale *et al*., 2020).

Habitat regulation

Seaweed canopies in the intertidal buffer the effects of high temperatures and water loss on organisms below their fronds, in particular when exposed to air at low tide. Below the canopy the substratum is much cooler than bare rock (Bertness *et al.,* 1999).

Nursery provision

A nursery can be defined as a habitat that contributes more than the average, compared with other habitats, to the production of individuals of a particular species that recruit to adult populations (Beck *et al*., 2001). Kelp forests as subtidal habitats form the primary habitat for many commercial and recreational fisheries that include a wide diversity of molluscs, crustaceans and finfish (Seitz *et al*., 2014, Laffoley & Grimsditch 2009 and references therein). For example, juveniles of commercially important crab species (e.g. brown crab, *Cancer pagurus* and velvet swimmer crab, *Necora puber*) have been recorded within both kelp holdfasts and understory algal assemblages in UK kelp forests (Teagle *et al*., 2018, Bué *et al*., 2020). Commercially important fish are associated with *A. nodosum* (i.e. Saithe: *Pollachius virens*, Atlantic cod: *Gadus morhua* and Atlantic herring: *Chupea harengus*), along with crustaceans (Pereira *et al.,* 2020).

Wave dampening, sediment retention (coastal protection)

Kelp beds can cause significant wave damping and the degree of wave breaking is reduced. It was also found that the kelp modifies the water velocity profile (Løvås & Tørum, 2001; Jackson 1997). As a result, currents should have different properties in the region of a kelp bed than in a similar kelp free region (Jackson 1997). Wave attenuation by kelp forests in shallow waters has been substantiated by measurements at Hustadvika at a site which is strongly exposed to waves from the open ocean. The reduction of wave energy from the outer to inner part of kelp belt over a distance of 258 m was 70-85 % with highest values at low tide when water depth over the bed is lowest (Mork 1996).

Seaweed beds also support erosion control as reduction in current velocities both within and adjacent to the beds can result in increased sedimentation and reduced turbidity (Madsen *et al*., 2001). Changes in hydrodynamics caused by seaweed may alter the supply and dispersal of algal propagules and invertebrate larvae, thereby affecting settlement processes (Teagle *et al*., 2017 and references therein).

The level of protection provided by seaweed varies seasonally, particularly during winter months, when loss of annual species or reduction of blades in perennial species reduces the amount of biomass in the water column (Christianen *et al*., 2013 cited from Scottish Government 2016).

Smaller seaweeds such as red and green species are less robust and create less structurally complex habitats, so that contribution to this ecological service is lower.

Waste remediation

Seaweeds are frequently used as indicator organisms in environmental monitoring, particularly in relation to heavy metals due to ability to bio accumulate contaminants (Gundersen *et al*., 2016 and references therein). In general, adult fucoid algae accumulate heavy metals and are generally fairly robust in the face of chemical pollution although, germlings appear to be intolerant of heavy metal pollution. However, local variation exists in the tolerance to copper. Plants from highly copper polluted areas can be very tolerant, while those from unpolluted areas suffer significantly reduced growth rates at 25 micrograms per litre (Eftec 2014 and references therein).

Leisure and recreation

While not a measure of ecological importance, kelp beds also benefit people by providing opportunities for leisure and recreation. Gundersen *et al*. (2016) indicate that kelp and mussel beds in Norway attract snorkelers and scuba divers and mitigate eutrophication improving dive site quality. In the UK also kelp beds are of importance for recreational divers and anglers, contributing to an estimated value of £11.7 billion for the UK alone (Beaumont *et al*., 2006, cited from Salomidi *et al*., 2012; Laffoley & Grimsditch 2009).

Cultural heritage

Porphyra spp. (laver bread) are the basis of a traditional dish produced in Wales since mediaeval times (O’Connor 2009). Recreational and commercial harvesting for this product continues today. Harvesting of edible seaweeds also provides recreational opportunities and interest for foragers.

2.3.1 Beach cast seaweed

Detached, beach cast seaweed transported onshore by storms and wave action also provides important ecosystem services (Orr, 2013). As this material becomes a refuge from desiccation and predation for intertidal organisms (Lewis *et al.,* 2007), it also forms an important feeding area for seabirds on poorly enriched beach habitats. Beach cast also provides coastal protection by aiding in the formation of sand dunes.

Beaches with wrack were associated with enriched benthic infauna (polychaetes) on the lower shore. While rack mounds supported abundant macroinvertebrates (mainly Diptera larvae and oligochaetes). These fauna are valuable prey to shorebirds, as demonstrated by a strong positive relationship between wader abundances and the percentage cover of wrack on beaches (Orr, 2013). The volume of drifting macroalgae inshore was a significant predictor (along with physical beach characteristics) for the abundance of decapods and fish (Orr, 2013).

3. Hand harvesting: methods

3.1 Methods and equipment

Both commercial and recreational hand harvesting are most commonly accomplished by selective cutting or sometimes plucking or pulling by hand. A mix of species may be harvested at the same time by recreational harvesters, whereas commercial harvest tends to be monospecific, focussed on a few target species which vary seasonally. The technique, equipment and yield per unit effort vary with target species, local conditions, access, and tradition (Mac Monagail & Morrison, 2020). Use of boats, rakes and diver-harvest has been reported to generate higher income than hand harvest from shore (Rebours *et al*., 2014).

Shore-based harvesting

Generally in the UK, knives, scissors, bill hooks, implements known as 'sickles' and serrated scythes are used for hand cutting intertidal seaweeds such as *P. palmata*, *A. nodosum, H. elongata*, fucoids and some kelps, with harvesting sites accessed on foot at low tide. Small red seaweeds including *M. stellatus* and *C. crispus* and greens Ulva spp. are harvested by plucking or using small knives, or scissors (Mac Monagail & Morrison 2020). No evidence was found on the extent to which seaweeds are hand plucked or pulled in Wales and England, although it is assumed to be restricted to recreational collection, with most commercial scale operators following guidance by selectively cutting using scissors or knives (pers. comm to Wilding 2021, Cornish Seaweed Co.). Harvested seaweeds are collected into bags, sacks or buckets for transport, either by boat or by road.

In Ireland, the 3rd largest European producer of *A. nodosum* (FAO, 2018), the wrack is commercially hand harvested with a sickle or small knife then tied with ropes into large bundles of 2-4 tonnes which are then towed away by boat after being allowed to float on the incoming tide (Mac Monagail & Morrison 2020). Other species are not suited to this method, as they either don't float or are not found at a convenient height on the shore.

Diving and snorkelling

Some species found lower on the shore, such as the kelps and *H. elongata*, may be harvested by free diving (Cornish Seaweed Co. pers com 2021). In Ireland and Scotland some companies harvest using diving apparatus, for example small volumes of *S. latissima* are diver-harvested as an aromatic in Harris Gin (McLaughlin *et al*. 2006; Angus, 2017).

Dislodgment

L. digitata is traditionally hand harvested in France during autumn, when whole plants are easier to dislodge from the bedrock due to a seasonal reduction in adherence of the holdfast (Pérez. 1971; Arzel 1989). However, this practice is not known to take place in Wales or England.

Boat and rake harvesting

Historically in Ireland, *A. nodosum* was harvested from boats using a crisín – a pole with a hook and crosspiece (Mac Monagail & Morrison, 2020). Boat and rake harvesting is now employed in some parts of Ireland, Scotland (in the Outer Hebrides by the company Uist Asco), Canada (Nova Scotia, New Brunswick and Prince Edward Islands), and the USA (Maine) to harvest *A. nodosum*, kelp and Irish moss *C. crispus* and *M. stellatus* (Mac Monagail *et al*., 2017). For *A. nodosum*, an open-hull boat is operated over the seaweed beds at mid-to-high tide, from which rakes are used to hand harvest. The rakes are specially designed, featuring a long handle and serrated cutting head with steel guards, and are lowered over the side then pulled through the floating seaweed canopy (Mac Monagail *et al*. 2017). Large clumps of seaweed are removed from the top of the canopy, generally leaving behind the tissue near the base of the seaweed, which allows for regrowth. Using this method, rake harvesters can land between 3 to 5 tons of *A. nodosum* per tide (Mac Monagail *et al*., 2017). While *A. nodosum* is harvested in Wales and England, it is thought that current methods are exclusively shore-based.

It should be noted that "drag-raking" is a mechanised process which differs from hand raking; drag rakes are small dredges towed behind a boat. Raking can also generally refer to mechanised raking; the Hebridean Seaweed Company in Scotland utilise a combination of hand and mechanical raking methods for *A. nodosum* harvest (Burrows *et al*., 2010).

Beach Cast "gathering"

The collection of detached, beach-cast, or drift seaweed is referred to as "gathering" (Angus 2017). While some beach cast weed may be gathered opportunistically on a small scale in Wales and parts of England for use as fertilizer, it is not known to be commercially collected, although may be removed from tourist beaches for management reasons (Perry *et al*., 2014). The practice of gathering beach cast weed is more common in Ireland, where washed up kelp stipes are known as "sea sticks" (McLaughlin *et al*., 2006), or Scotland, where the activity and forms a key aspect of "crofting" (Angus 2017).

This activity is likely to be seasonal, following winter storm events in which large volumes of seaweed are washed ashore. Motivation for collection of drift weed includes for recreational or commercial use as a fertilizer (McLaughlin *et al*., 2006), animal feed (Guiry& Morrison 2013), or simply to remove the seaweed deposits for aesthetic reasons, particularly on popular tourist beaches, because the washed in material quickly begins to decompose and become considered a nuisance.

A mixture of species are likely to compose the drift weed collected in Wales and England . This will vary regionally, but is likely to be dominated by kelps (Laminaria spp. and sometimes *S. latissima*) and to a lesser extent brown seaweeds (McLaughlin *et al*., 2006; Perry *et al*., 2014). In the western Atlantic, drift weed is referred to as "storm toss" and is dominated by *C. crispus* and *Furcellaria lumbricalis* (FAO.org)

The materials used for gathering of beach cast weed are various, and may include a variety of hand rakes, forks and spades, buckets, bags, wheelbarrows or hand-carts, nets, horses, and larger vehicles used for transport (Mac Monagail *et al*., 2017). Removal using vehicles including tractors, bulldozers and lorries is common (McLaughlin *et al*., 2006), but is considered to be mechanical harvesting and as such is beyond the scope of this report.

3.2 Seasonality of harvest

Optimal harvesting periods are species specific and influenced by a range of factors including growing seasons (see 2.3 on growth rates; Appendix 15 and species appendices for key information). The kelp *A. esculenta,* for example, has a narrow widow for harvest in late spring between reaching peak growth and erosion (Scottish Government 2016). *Ulva* spp. are mainly targeted in summer while *Fucus* spp. may be harvested year-round on an ad-hoc basis (Cornish Seaweed Co. pers com 2021). A review of commercial and recreational seaweed harvesting in Wales, Perry *et al*. (2014) found that harvesting activity for *P. palmata* and *Porphyra* spp. takes place in both winter and summer, while kelps are collected mainly in summer.

Other factors may also influence choice of harvesting period. The chemical composition of seaweeds varies seasonally (Marinho *et al*., 2015; Mols-Mortensen *et al.,* 2017; Schiener *et al*., 2015), so the desired proportion of bioactives, protein, carbohydrate or flavour required for the end use will also influence harvesting season. *H. elongata* in the UK is commercially harvested year round, but is most palatable in early summer (Angus 2017) and reaches maximum biomass through summer.

3.3 Comparison of different harvesting methods

The sustainability of seaweed resources will be influenced by the appropriateness of tools used to harvest, although the magnitude of the impact will also be dependent on the intensity and frequency of harvest, species exploited, and local environmental conditions (Mac Monagail *et al*., 2017). Intuitively, plucking or pulling could potentially be more damaging to smaller species, such as *Osmundea* spp. or *Ulva* spp. due to possibility of uprooting the entire plant. Very few controlled studies compare the impact of different harvesting techniques according to hand harvesting method or species, so it is difficult to draw conclusions about which method has a greater or lesser impact.

The relative impact of hand harvesting with mechanised methods is also difficult to directly compare and can require case specific consideration. Suction-based mechanical harvesting of *C. crispus* and *A. nodosum* was banned in Canada in the 1990's due to overharvesting only 8 years after its introduction (Ugarte & Sharp 2012). It was replaced by the more traditional boat and hand rake method (outlined above), which appears to be both sustainable and high yielding for *A. nodosum*, removing large amounts of biomass from the tips with an effect of stimulating dense, bushy growth, similar to pruning of terrestrial crops (Mac Monagail *et al*., 2017). Recovery of the canopy occurs in 1-2 years (Sharp *et al*., 2006).

Further, a study from Brittany comparing hand harvesting of *L. digitata* at low tide with the use of a mechanised "scoubidou trawl" found no difference in recovery times after harvest (Perez 1969).

Mechanical harvesting has either been restricted or met with resistance in Ireland, Northern Ireland, Scotland and the Basque, on environmental grounds and risk of impact to Natura 2000 protected sites.

Perhaps more critical to recovery of the seaweed canopy is the amount of the seaweed plant remaining intact following harvesting, rather than specifically the method (Seip 1980, see recovery section above). Kelly *et al*. (2001) directly compared boat based hand (which left 20cm) and mechanical (which left 50cm) methods, for *A. nodosum* in Ireland. With both, biomass was fully restored within 18 months, and it was suggested that mechanical harvesting had less of an environmental impact than hand harvesting at a local, short term scale, due to the greater amount of the algae left remaining (Kelly *et al*. 2001). This is supported by research from Maine, USA, that found no evidence of a difference in impact between mechanical and hand-harvest when cutting height is regulated (Phillippi *et al*. 2014).

3.4 Ecological implications of harvesting

Removal of seaweed by hand harvesting is likely to have a range of ecological implications. The level of impact will depend on the intensity of harvesting, the spatial scale and the frequency of harvesting events which will also influence the recovery rate. Overall, harvesting will reduce the contribution of targeted seaweeds to ecosystem processes and functions. The following sections discuss the type of impacts and loosely follows the structure of the ecological importance section. Species specific information is presented in Appendices 1-14.

Population level effects

Seaweed harvesting can alter the age and size structure of targeted populations for some species. These changes may have ramifications for reproductive capacity and recovery.

In response to repeated annual mechanised harvesting in Brittany, the average age of *L. digitata* plants in the canopy decreased to <3 years old (Arzel, 1989 in Werner & Kraan 2004). *L. digitata* are fertile in their second year but their reproductive capacity/potential is greatest at 4 years old. Therefore, lowering the average age of canopy plants to <3 years may negatively impact recruitment in the long-term (Werner & Kraan 2004).

Steen *et al*. (2016) found that, while juveniles can quickly replace harvested adults in pristine kelp beds, the density of juvenile *L. hyperborea* in recovering areas was lower than in unharvested beds. As such, repeated harvesting may reduce the density of juveniles and the capacity of the kelp bed for recovery. Additionally, Christie *et al*. (1998) found that repeated clearing at 5-6 year intervals, results in the development of a very dense, homogenous kelp forest with lower species diversity.

Harvesting has also been reported to impact the size-class structure of the red seaweed *C. crispus* population by reducing mean frond length (McLaughlin *et al* 2006). Chopin *et al*. (1988) and Sharp and Pringle (1990) both reported that harvested beds of *C. crispus* showed reduced reproductive capacity than harvested beds.

Conversely, *P. palmata* appears to be more resistant to (careful) harvesting. Garbary *et al* (2012) found that while there was little difference in average cover of *P. palmata* between harvest and non-harvest shores, frond length was greater and frond density was three times higher on harvested shores than on the non-harvested shores.

Broadly, small scale, artisanal, hand harvest of seaweeds is considered to be a relatively benign method believed to be sustainable provided that certain guidance is followed (Angus, 2017; Scottish Government 2016). Despite this, hand harvest can result in localised depletion of some stocks if not managed or regulated. For example, sea-spaghetti *H. elongata* which is a popular food has reduced in abundance in some areas of France and Portugal where unregulated harvesting increased 35% between 2009 and 2013 (Mac Monagail *et al*., 2017).

Concern has also been raised about the sustainability of hand harvesting of small red and green seaweeds in the UK (Perry *et al*., 2014; Scottish Government 2016). Harvesting, which targets plants >5cm, was found to impact blade length of *P. dioica* resulting in a 64% reduction in blade size compared to unharvested areas. Hand harvesting was found to have minimal impact on percentage cover, yield, and associated community, while natural variation due to disturbance (sand cover) was high (Knoop, 2019). Although small scale, the future of unregulated seaweed harvesting activity in Wales, particularly for Laver (especially at hotspot sites in Pembrokeshire) and fertilizer, has been identified as a concern (Perry *et al*., 2014).

Removal of non-target species (by-catch)

Epiflora, hydroids, bryozoans and other encrusting fauna may be removed from the substratum along with seaweeds and are unlikely to re-attach and will die. Harvesting management and guidance should consider the risk of bycatch by careless harvesting for species of conservation interest. Examples of species potentially at risk are the seaweeds *Padina pavonica* (Peacock's tail) and *Anotrichium barbatum* (Bearded red seaweed) and the stalked jellyfish *Haliclystus auricula* and *Lucernariopsis campanulata*. Stalked jellyfish attach to seaweeds and seagrasses and as these are small they could be overlooked by unaware harvesters.

Careful harvesting will minimise by-catch although it is unavoidable for species that are attached the seaweed itself. While some species, such as grazing littorinids and crabs may fall off during harvesting, species that are attached more securely may have to be removed by hand. Where this processing is done on the shore some of these may relocate and survive although few epifauna and epiphytes are likely to be able to reattach. Processing away from the shore will remove by-catch from the ecosystem, which may be desirable for invasive non-native species but suboptimal for native species. Actual rates of by-catch on harvested seaweed are unknown.

Primary production and nutrient cycling

Stagnol *et al*. (2013) investigated the effects of commercial harvesting of intertidal *F. serratus* on ecosystem biodiversity and functioning. The study found that the removal of macroalgae affected the metabolic flux of the area. Flows from primary production and community respiration were lower on the impacted area as the removal of the canopy caused changes in temperature and humidity conditions.

As detritus can be transported over long distances this effect is not confined to the immediate habitat. System level effects are challenging to assess as the role of seaweeds and the transport and fate of organic carbon, especially from kelps is still being investigated. Analysis of data for a single year from 10 sites around Nova Scotia suggested that the removal of biomass of *A. nodosum* from coastal environments by harvesting was associated with a reduction in the amount of detrital material entering the food web (Halat *et al*. 2015). However, the actual amount and impact has been debated (Garbary *et al*. 2017; Ugarte *et al*. 2017).

Secondary production

The removal of seaweed will remove carbon and other nutrients from the ecosystem that would otherwise have supported grazers and provided organic detritus to support suspension feeders and underpin marine food chains.

Loss of associated species can directly impact predators but also result in trophic effects for top predators such as mammals that feed on fish. In Norway it was found that birds performed significantly more dives in kelp-forested areas versus cleared areas suggesting that removal of kelp habitat is associated with a reduction in bird foraging efficiency (Lorentsen *et al*. 2010). The potential impact of changes in the density and overall extent of kelp forests on fishery yields is still poorly known but most studies show a positive relationship between kelps and fish populations (Bertocci *et al*. 2015). Significant quantities of *A. nodosum* and other species of macroalgae were harvested from Strangford Lough, with possible effects on the local ecosystem. In 1744, Harris suggested that one of the reasons for herring stock decline in Strangford may have been kelp burning during the spawning season; the removal of seaweed in sheltered bays may also have contributed to declining numbers (McErlean *et al*. 2002).

Carbon sequestration

Seaweed harvesting will directly remove the carbon stored in the tissues. No direct evidence was found to assess changes in carbon storage and sequestration resulting from harvesting. The role of seaweeds (particularly highly productive kelps) in carbon storage has been largely overlooked until relatively recently. It is entirely unknown how seaweed harvesting would affect carbon cycling in coastal waters, but intuitively it is likely to reduce total carbon fixed and donated, at least at local/regional scales.

Habitat provision

Removal of seaweed will result in direct losses of attached epiphytes and epifauna and reduce habitat suitability for understorey fauna. Epiphytes are an important secondary habitat to many invertebrates such as amphipods (Christie *et al*. 2007) and a reduction in epiphytic abundance will slow the recovery of other fauna within cleared areas (Christie *et al*. 1998). Where holdfasts are left in place (in line with management recommendations, see Section 4.3), losses of holdfast fauna will be reduced although there may still be some changes in habitat conditions and suitability.

Impacts of harvesting on habitat provision are species specific. Harvesting of *H. elongata* had little impact on the associated community (Stagnol *et al*. 2015) and removal of the canopy encouraged development of germlings.

The rate at which different taxa re-colonise depends on their dispersal abilities and reproductive strategy and slow-moving fauna are more vulnerable to the impacts of seaweed removal. Studies on kelp have found that the most rapid colonisers are amphipods that are strong swimmers and gastropods that drift in the water column as planktonic larvae (Waage-Nielsen *et al*. 2003). Slower colonisers include sessile fauna (mussels) and those that reach new habitats by weak swimming and/or crawling such as polychaetes and isopods (Waage-Nielsen *et al*. 2003).

The removal of seaweeds will expose species to predation by birds and fish. Changes in *A. nodosum* have been found to affect the large, mobile fauna such as crabs or grazing gastropods (Bertness *et al*. 1999; Fegley 2001; Jenkins *et al*. 1999; Phillippi *et al*. 2014; Pocklington *et al*. 2018). Mobile species such as crabs and amphipods may rapidly relocate to suitable areas. However sessile and slow-moving species such as limpets and dog whelks may however be subject to increased predation.

Habitat modification

The removal of seaweed canopy exposes understorey species to sunlight and aerial conditions during low tides resulting in bleaching and eventual die backs. Experimental removal experiments conducted in the Isle of Man (Hawkins & Harkin, 1985) found that following the removal of the seaweed canopy the understorey encrusting red algae became bleached and died within a week. Removal of the mature kelp canopy, conversely allows more light to penetrate the understorey which stimulates rapid growth of the small kelp recruits (Christie *et al*., 1998) which generally persist in the understorey for several years).

Pocklington *et al*. (2018) examined community disturbance after removal of different proportions of the of the *A. nodosum* fronds (but not holdfasts). Removal of 50% of fronds increased the temperature under the canopy significantly and decreased the abundance of mobile invertebrates such as *Littorina obtusata*. Sessile taxa such as *Osmundea pinnatifida* and encrusting corallines could withstand a 75% loss of fronds but declined by half if 100% were removed.

Changes in community composition

Loss of seaweeds results in the direct loss of attached species and results in a reduction in habitat for associated species as described above. The removal of seaweeds may also allow the establishment of a different type of community. The removal of native seaweeds could provide opportunity for the establishment of non-native seaweeds. As non-native species are difficult to eradicate, their introduction may permanently change the character of a habitat having implications for those species which rely on seaweeds to provide habitat, shelter and food.

Clearing kelp typically results in outbreaks of understorey algae such as the opportunistic brown seaweeds *Desmarestia viridis* and *D. aculeata*, and the short-lived kelps, *A. esculenta* and *S. polyschides* (Kain 1975). In Brittany, where *L. digitata* is at its most southerly range, it is increasingly being outcompeted by the fast growing kelp *S. polyschides* after mechanised harvesting (Werner & Kraan 2004; Mac Monagail *et al*. 2017). This is consistent with research from clearance plots in the same region (Engelen *et al*. 2011). If hand harvesting causes the *L. digitata* canopy to become more fragmented and disturbed, it could lead to an increased abundance of *S. polyschides* (as has been observed in France) or facilitate the spread of the opportunist invasive species, *U. pinnatifida* and/or *L. ochroleuca* further into Wales and England.

Numerous studies have concluded that *A. nodosum* take long periods of time to recover from removal, with cleared areas being dominated by *Fucus* spp. (Bertness *et al*, 2002; Jenkins *et al*. 1999; Dudgeon & Petraitis 2005; Cervin *et al*. 2005; Ingólfsson & Hawkins 2008). Large clearings (~8m diameter) were found to be quickly colonised by competing species, whereas small clearings allowed *A. nodosum* to recover more effectively (Dudgeon & Petraitis 2005), presumably by processes of vegetative growth from base of plants remaining nearby. Stagnol *et al*. (2013) found that opportunistic ephemeral green algae such as *Ulva* spp. increased following removal of intertidal canopy forming seaweeds.

In some instances, seaweed beds are replaced by other similar species. In Canada, declines of harvested *C. crispus* resulted in replacement by another red seaweed, *Furcellaria lumbricalis* (Sharp *et al*. 1993). In other instances, harvesting by raking of *C. crispus* led to replacement by *Corallina officinalis* and encrusting coralline biotope (MacFarlane 1952). These changes appeared to be driven by increases in gastropods which prevented the recovery of *C. crispus* by grazing. Stagnol *et al*. (2013) observed another grazer, the limpet *Patella vulgata* recruiting in bare patches of disturbed plots. Limpets control the development of macroalgae by consuming microscopic phases (Jenkins *et al*., 2005) or the adult stages (Davies *et al*., 2007).

Large scale loss of fucoids can cause systems shifts to a state dominated by low-lying turf or filamentous ephemeral algae. Turf algae, especially corallines, are often highly resilient and positively associated with perturbed areas and can recover and reach greater abundance compared to prior disturbance conditions (Bulleri *et al*. 2002; Bertocci *et al*. 2010). These turf algae can then prevent canopy recovery by inhibiting recruitment.

Habitat changes in extent, fragmentation, connectivity

Clearing seaweeds can create bare areas. Attached and sedentary species will have limited or no ability to move to adjacent areas. The successful dispersal of mobile fauna between and within fragmented habitats will depend on the size of the cleared area as well as the dispersal abilities of fauna associated with the seaweed. Most fauna associated with kelp are able to disperse rapidly across cleared areas and tracks that are more than 10 m wide (Waage-Nielsen *et al*., 2003). A study in Norway found that 87% of mobile species within large cleared areas (~ 5000 m^2), were able to re-colonise suitable substrates (e.g. nearby kelp holdfasts) within 35 days (Waage-Nielsen *et al*., 2003).

Nursery provision

The removal of kelp habitat may trigger an immediate, localised reduction in juvenile fish due to the loss of shelter and food (Bodkin, 1988; Lorentsen *et al*., 2010). Juvenile fish within the cleared areas may become easy targets for predatory fish and birds (e.g. cormorants) so that small fish migrate to the nearest kelp-forested areas to seek refuge (Lorentsen *et al*., 2010). Where large quantities of *L. hyperborea* were removed (~15 000 metric tonnes) in Norway, the abundance of small (< 15 mm) gadoid fish was 92% lower in cleared areas versus kelp-forested areas, although abundances of larger fish (> 15 mm) were similar in cleared and forested areas (Lorentsen *et al*., 2010).

Wave dampening, sediment retention (coastal protection)

Kelp and fucoid canopy species can decrease wave energy, reducing coastline erosion and increasing sedimentation rates. Broadscale removal of canopies would reduce coastal protection, increasing natural hazard risks. Increased wave action and water currents may also reduce settlement of larvae and propagules, decreasing recruitment.

Habitat sensitivity to trampling

Access by vehicle or foot could damage areas of the shore used for seaweed harvesting. Tyler-Walters and Arnold (2008) reviewed the impacts of vehicle access and trampling on a range of intertidal habitats, that report should be referred to for sensitivity to vehicle access and the detailed information. Sensitivity to foot access was assessed as low, medium or high, based on the methodology developed by Hall *et al*., (2008). For all habitats that may contain targeted seaweeds, the sensitivity to trampling is summarised below in Table 3. Sensitivity was assessed for four levels of intensity, based on Hall *et al*., (2008) for access to fishing grounds:

- Single: access on a single occasion;
- Light: access by 1-2 people per hectare per day;
- Moderate: access by 3-9 people per hectare per day; and
- Heavy: access by >10 people per hectare per day or large numbers of individuals mainly concentrated in one area.

The review found that trampling studies and their results were highly variable and depend on the nature of the habitat. Trampling impacts resulted from physical contact and wear and were dependant on the intensity, duration, and frequency of trampling, and even the type of footwear used. Increased trampling intensities result in reduced biodiversity, reduced abundance or biomass of affected species (especially seaweeds) and increased bare space and, in some cases, clear paths were visible.

Brown seaweeds were particularly intolerant and sensitive to trampling impacts. Understorey algae could suffer indirectly if these were impacted due to increased desiccation but more robust algal turf species, opportunists and gastropod grazers (e.g. limpets) could increase in abundance as an indirect effect of trampling. Trampling damaged erect coralline turfs and resulted in an increase in bare space. In some cases, paths across the shore were visible.

Trampling may also impact habitats of conservation interest, those that are sensitive include honey comb worm reefs (*Sabellaria alveolata*) and seagrass beds.

A useful source of further information on physical damage pressures is the MarLIN website. While the sensitivity assessments presented for habitats (biotopes) do not specifically consider seaweed harvesting and trampling, the assessments for the pressures: abrasion, removal of target and non-target species are likely to be useful to managers.

Table 3. Summary of impacts from trampling by foot at different intensities based on Tyler-Walters and Arnold (2008). Sensitivity was assessed as low, medium or high, based on the methodology developed by Hall *et al*., (2008).

Habitat	**High intensity**	**Medium intensity**	**Low intensity**	**Single**
Upper shore stable rock with lichens and algal crusts	High	High	Medium	Low
Wave exposed stable rock	High	Medium	Medium	Low
Moderately wave exposed rock	High	High	Medium	Low
Brown and red seaweeds and mussels on moderately exposed lower shore rock	High	High	Medium	Low
Mussels & boring bivalves (piddocks) on clay and peat	Medium	Medium	Medium	Low
Honey comb worm reefs	High	Medium	Medium	Low
Sheltered bedrock, boulders, cobbles	Medium	Medium	Medium	Low
Rockpools and overhangs on rocky shores	Medium	Medium	Medium	Low
Brown seaweeds, barnacles or ephemeral seaweeds on boulders. Cobbles and pebbles	Medium	Medium	Medium	Low
Saltmarsh	Medium	Medium	Medium	Low
Underboulder communities on lower shore and shallow sublittoral boulders and cobbles	Medium	Medium	Medium	Low
Biogenic reef on sediment	Medium	Medium	Medium	Low
Seagrass beds	High	Medium	Medium	Low

Impact of gathering drift / beach - cast weed

In Scotland, beach-cast kelp provides an essential food source to beach invertebrates and has been shown to play a vital role in coastal food webs (Orr 2013). The biomass of fly larvae found in mounds of rotting seaweed in the Uists are some of the highest reported globally, and these larvae are a critical food source for shorebirds that stop-over on the islands to ‘refuel’ during their spring and autumn migration passages (Orr 2013).

Seaweed decaying on beaches or on the seabed is broken down and re-mineralized (e.g. by microbial activity and invertebrate grazers), and the nutrients are exported to the nearshore environment (Revell *et al*. 2011). The process of nutrient recycling is broadly recognized as being essential in maintaining ecosystem functioning, by facilitating the growth of primary producers such as phytoplankton and kelp (Soares *et al*. 1997; Raffaelli 2006; Bulling *et al*., 2010). Removal of beach-cast weed is likely to impact on these functions.

3.5 Hand harvesting and non-native species

Species which have been introduced into areas outside their natural range through human actions and are posing a threat to native wildlife and ecosystems, are known as invasive non-native species (INNS). Shipping (hull fouling and ballast water) and transport of aquaculture stock, such as oysters, are the main pathways by which marine INNS are transported long distances. Species may also be introduced intentionally, typically as commercially cultivated species. The Japanese seaweed, Wakame (*U. pinnatifida*), for example) was introduced to Brittany as a cultivated species. Once a species has become established, further dispersal may take place either naturally through the dispersal of reproductive propagules and fragments capable of regenerating, drifting or through attachment to other species. Human activities such as movement of fouled objects, e.g. commercial and recreational boating can also disperse INNS. Hand harvesting of seaweeds may result in unintended movement of INNS which may increase spread. Some movement of INNS from harvesting may be similar to other activities that involve movement and disturbance, where adults or propagules are carried to other areas on objects or clothing.

To identify marine INNS that might be spread by hand harvesting of seaweeds and to prioritise these for management a risk assessment exercise was undertaken. This identified species of concern, the ways in which these spread, their level of impact on the environment and the degree to which these have already spread in the UK.

Much of the supporting information for the risk assessment is presented in Appendices, 18-20. Seaweed non-natives were also identified in a separate checklist (Appendix 18) to ensure that managers are aware these are non-natives.

Marine invasive non-native species present in the UK

A high level screening exercise was undertaken with INNS experts at the MBA to determine which non-natives are likely to be associated with harvested seaweed and/or their habitats. A list of approximately 90 INNS species was generated that are associated with intertidal and subtidal habitats. From this initial full list, 30 INNS algae that may be suggested for harvesting were identified (see Appendix 18).

To prioritise species, a rapid assessment was made of whether establishment of the species is likely to lead to significant ecological impacts on other species and habitats. The assessment was based on information from native and invaded ranges, and used a number of key sources, including the GB non-native species secretariat (online) CABI online invasive species compendium Smithsonian Environmental Research Center's online National Estuarine and Marine Exotic Species Information System (NEMESIS) and a previous report for NRW by Tillin *et al*. (2020). The sift for ecological impact resulted in a reduced list of 63 species which excluded species considered to be low risk in terms of ecological impacts.

For species that are potentially of concern based on impact, a rapid review was undertaken to identify the likely habitats of each INNS to assess if these may occur in areas with seaweeds. INNS were categorised into three classes, invasive seaweeds, attached/fouling species and sheltering mobile species (see Table 4). The modes of local/regional dispersal, current distribution in the UK and key evidence sources are presented in Appendix 19.

Table 4. INNS species that may be found in the same habitats as seaweeds and which can lead to high impacts on native habitats and species.

Invasive seaweeds	**Attached/fouling species**	**Sheltering mobile species**
Agarophyton vermiculophyllum	*Asterocarpa humilis*	*Ammothea hilgendorfi*
Asparagopsis armata	*Botrylloides diegensis*	*Boccardia proboscidea*
Bonnemaisonia hamifera	*Botrylloides violaceus*	*Caprella mutica*
Caulacanthus okamurae	*Bugula neritina*	*Cephalothrix simula*
Codium fragile subsp. fragile	*Ciona robusta*	*Hemigrapsus sanguineus*
Colpomenia peregrina	*Cordylophora caspia*	*Hemigrapsus takanoi*
Dasysiphonia japonica	*Corella eumyota*	*Palaemon macrodactylus*
Grateloupia subpectinata	*Didemnum vexillum*	*Urosalpinx cinerea*
Grateloupia turuturu	*Hydroides ezoensis*	Blank cell
Melanothamnus harveyi	*Schizoporella japonica*	Blank cell
Pikea californica	*Styela clava*	Blank cell
Sargassum muticum	*Tricellaria inopinata*	Blank cell
Solieria chordalis	*Watersipora subatra*	Blank cell
Undaria pinnatifida	Blank cell	Blank cell

Risk of spread of invasive non-natives

For each of the INNS, information was gathered on modes of dispersal to understand how the species might spread, this information is provided in Appendix 19. For most species, information was relatively limited and no attempt was made to weight different pathways according to which are of more importance. Summary tables identify if the species is naturally dispersed by propagules in the water column, whether it can drift or whether it is transported attached to other species or objects, whether regeneration from fragments is possible and if the species is mobile (see Appendix 19). Invasive species typically have traits that support dispersal and all the assessed species are able to disperse using more than one pathway.

Species that are mobile or that can disperse over long distances through propagules are less likely to be contained by management objectives and these paths are not considered relevant to hand harvesting of seaweeds. Harvesting however may increase dispersal by detaching or dislodging species that can drift and through movement of species or objects with attached non-natives. If detached or transported species can regenerate from fragments this increases likelihood of dispersal.

Each INNS species was ranked according to risk of spread from harvesting according to the following categories:

High relevance: species can drift and regenerate from fragments and are spread through attachment to species or objects but have low or limited natural dispersal and are not mobile.

Medium relevance: species are spread through attachment to species or objects, have low or limited natural dispersal through propagules and the species does not regenerate from fragments.

Low relevance: species can spread through attachment to species or objects but also can disperse naturally through propagules and/or are mobile.

Current distribution of invasive non-native species was assessed using records in the National Biodiversity Network online Atlas. This is not complete but provides a readily available source. Species were labelled as either widespread where there are numerous records over a wide area and restricted where there were fewer records and in smaller areas. Brief details are provided in Appendix 19. Although this assessment is based on the most recent and reliable information it should be treated with some caution as species may be under recorded and not all records are reported to this resource.

Impact assessments for invasive non-native species

The potential impact pathways of INNS on ecosystems are summarised below, adopting the categories developed for the Environmental Impact Classification of Alien Taxa (EICAT) as this list is comprehensive. The pathways are:

- Species level impacts
    - Competition
    - Predation
    - Hybridisation
    - Transmission of disease or parasites
    - Parasitism This impact mechanism is restricted to species that are parasites
    - Poisoning/ toxicity
    - Bio-fouling or other direct physical disturbance
    - Grazing/ herbivory/ browsing
    - Indirect impacts through interaction with other species
- Chemical impact on ecosystems
- Physical impact on ecosystems.
- Structural impact on ecosystems

Supporting information using this categorisation is presented in Appendix 20. EICAT provides guidance for assessing the severity of impacts as shown below in Table 5. For each INNS an assessment was made of the likely level of impact.

Table 5 Impact categories and definitions adopted from the EICAT risk assessment methodology. Habitat impact qualifiers (in italics) are based on a previous project by Tillin *et al*., (2020).

Impact category	**Definition for impact on native species or habitat features**
Massive	Irreversible local, or global extinction of a native taxon (i.e. change in community structure) *and/or irreversible change to habitat character, e.g. loss of biogenic habitat or substratum type change, e.g. sediment to biogenic habitat structured by INNS.*
Major	Native taxon local extinction (i.e. change in community structure), *and/or change to habitat character, e.g. loss of biogenic habitat or substratum type change, e.g. sediment to biogenic habitat structured by INNS which is reversible.*

Moderate	Native taxon population decline *and/or alteration to key habitat features but habitat is still recognisable.*
Minor	Performance of individuals reduced, but no decrease in population size *and/or some alteration to habitat but not to degree that would impact key characterising species or habitat categorisation, structure or functioning.*
Minimal Concern	Negligible impacts, and no reduction in performance of native taxas' individuals, *negligible impacts on habitat.*
Data deficient	No evidence to assess.

The assessment shows (Appendix 20) that the predominant pathways through which the INNS impact ecosystems are through competition and biofouling and biofouling associated structural changes in habitats. There was no supporting information for hybridisation, transmission of disease or parasites, or chemical impacts. Species which are considered likely to lead to the most significant impacts on natural habitats are the invasive seaweeds: *Asparagopsis armata, Dasysiphonia japonica, Sargassum muticum* and *U. pinnatifida*, the tunicates, *Botrylloides diegensis*, *B. violaceus* and *Didemnum vexillum*.

Species that were assessed as having moderate impacts reduce the populations of native species but do not result in significant changes to habitats. These include seaweeds which may compete for space and light with native seaweeds, including *Agarophyton vermiculophyllum*, *Bonnemaisonia hamifera*, *Caulacanthus okamurae*, *Daysiphonia japonica, Grateloupia subpectinata, G. turuturu*, *Melanothamnus harveyi* and *Solieria chordalis*. Attached and fouling species compete with native fouling communities and may alter habitat structure, this group includes the hydroid *Cordylophora caspia*, bryozoans *Bugula neritina*, *Tricellaria inopinata* and the tubeworm *Hydroides ezoensis*. Predatory species, *Urosalpinx cinerea*, *Hemigrapsus sanguineus* and *H. takanoi* may reduce abundance of prey species, with *U. cinerea* of particular concern due to potential impacts on bivalves of commercial and conservation interest such as mussels and oysters (Tillin *et al*., 2020).

Although human health and socio economic concerns were not the focus of the assessment it was noted that the bryozoan, *Tricellaria inopinata* and other fouling species may reduce the commercial value of hand harvested seaweeds where these encrust. Kelps in particular can become heavily fouled by epifauna and epiphytes, particularly in sheltered sites. As well as reducing productivity, processing times may be increased where fouling species have to be removed.

The only species of concern regarding toxicity was the nemertean (ribbon worm) *Cephalothrix simula*. Individuals collected in the UK have contained the neurotoxin Tetrodotoxin (TTX) and analogues of it (Turner *et al*., 2018). TTX is a potent neurotoxin that is responsible for countless human intoxications and deaths around the world and is found in a wide range of marine phyla. Asakawa *et al*. (2013) reported that around 80% of specimens of *C. simula* collected in Japan were toxic, whereas other nemerteans generally were not. The risk from TTX-positive nemerteans entering the food chain therefore needs careful assessment, although worldwide no incidents of TTX poisoning have been directly attributed to *C. simula* (Turner *et al*. 2018).

Priority species for management of invasive non-native species risks

Species priority for management was assessed based on impact, current distribution and relevance of seaweed harvesting for spread (see Appendix 21 for full output table). Species were prioritised for management concern, based on the following categories.

1: Ecological impact: moderate, major, or massive, relevance of seaweed harvesting for spread is high and current distribution is restricted. *C simula* was added as a priority species given concerns over toxicity.

2. Ecological impact: moderate, major, or massive, relevance of seaweed harvesting for spread is high and current distribution is widespread. *U. pinnatifida* was added to this group, given its relevance to harvested seaweed.

3. Ecological impact: moderate or greater, distribution is restricted, relevance of seaweed harvesting for spread is medium.

4 . Ecological impact: moderate or greater, distribution is widespread, relevance of seaweed harvesting for spread is medium.

5 All other species.

Table 6 below shows the highest priority species for management. These are species which result in the greatest ecological impacts and which may be spread by seaweed harvesting. Those that are highest priority have restricted distributions and therefore constraining dispersal is a clear management goal. For species that are widespread (Priority 2), natural dispersal is limited and reducing further dispersal should be considered desirable. Both widespread and more spatially restricted species were prioritised due to caveats around assessing distribution from the NBN atlas and as species records are biased towards more surveyed areas, with some species being difficult to find or identify and under reported.

Fourteen INNS were prioritised for management concern. These were mostly invasive seaweeds (twelve species) and the tunicate *Didemnum vexillum* and the nemertean *C. simula* (based on health concerns).

Table 6. Highest priority invasive non-native species for management.

Species name	**EICAT (Impact)**	**Distribution**	**Relevance**	**Priority**
Agarophyton vermiculophyllum	Moderate	Restricted: West coast only	High	1
Asparagopsis armata	Major	Widespread: West and south coasts	High	2
Bonnemaisonia hamifera	Moderate	Widespread: West and south coasts	High	2
Caulacanthus okamurae	Moderate	Restricted: West and south coast	High	1
Dasysiphonia japonica	Massive	Restricted: Widespread on west coast only	High	1
Grateloupia subpectinata	Moderate	Restricted: South coast	High	1

Species name	EICAT (Impact)	Distribution	Relevance	Priority
Grateloupia turuturu	Moderate	Widespread: South coast	High	2
Melanothamnus harveyi	Moderate	Widespread: Mainly west and south coasts	High	2
S. muticum	Major	Widespread	High	2
Solieria chordalis	Moderate	Restricted: Mainly south coast	High	1
U. pinnatifida	Major	Widespread	Low	2
Cordylophora caspia	Moderate	Widespread	High	2
Didemnum vexillum	Major	Restricted: West and south coast	High	1
Cephalothrix simula	Moderate	No records (NBN)	Medium	1

Harvesting of invasive non-native species

Harvesting of invasive non-native species is unlikely to be permitted but this section outlines some considerations around harvesting based on *S. muticum* as it is likely that some recreational hand harvesting of invasive species takes place and proposals may be submitted for control by harvesting. Hand harvesting may be considered desirable in terms of limiting the spread of non-natives by removing these from the wild and reducing the size of their population and therefore reproductive capacity. However, it is an offence to spread certain non-native species and to plant or otherwise cause to grow in the wild any plant listed in Schedule 9 of the Wildlife and Countryside Act (1981), and this currently applies to two species in Wales, *Sargassum muticum* and *U. pinnatifida*. While the sale of harvested *U. pinnatifida* is currently not permitted under Schedule 9 of the Wildlife and Countryside Act, it is already harvested in parts of England for personal use. This would include replacing or disposing of certain seaweeds to the sea, even if they were put back at the site where they were collected.

The main invasive non-native seaweeds likely to be of interest for commercial harvesting are *S. muticum*, *U. pinnatifida* and *Asparagopsis armata* (harpoon weed) (see Appendix 18). *A. armata* was introduced to Europe as a commercially cultivated species. However, no evidence was found for commercial or recreational collection in the UK.

S. muticum, a north western Pacific brown alga is a successful invader because it is fast growing and has high reproductive potential, is tolerant to a wide range of salinities, and its air bladders allow dispersal by drifting for dislodged plants or fragments. It has been described as the most ‘successful’ invasive species in the UK in terms of its rate of spread (Davison 2009). The harvesting of *S. muticum* has failed to eradicate it from the coast of the British Isles, but regular harvesting is used as a method to reduce its spread and the problems caused by its growth (Critchley *et al*. 1986; Lodeiro *et al*. 2004).

Critchley *et al*. (1986) reviewed the methods and cost of harvesting *S. muticum*. The harvesting of *S. muticum*, in an attempt to control it, results in the need to dispose of large quantities of seaweed biomass (Davison 2009). Re-growth following harvesting was often more dense and vigorous than the previous population because the removal of the algal canopy and the creation of patches of bare substrate were favourable to *S. muticum,* which can rapidly colonise bare spaces. It has also been suggested that such attempts could facilitate dispersal by creating large quantities of drift *S. muticum* fragments (Fletcher & Fletcher, 1975). A further harvesting consideration is that stalked jellyfish, that are of conservation interest have been found on *S. muticum* (pers. comm. to report authors by Liz Bailey, Natural England).

Although, *S. muticum* has been exploited for aquaculture in China (Liu *et al*. 2013) and as a traditional food in Korea (Yang *et al*. 2013), there is currently no commercial exploitation of this biomass in Europe (Lodeiro *et al*. 2004). The utilisation of *S. muticum* biomass for fuel and other products could encourage its harvesting and control and eradication may be successful in small areas but would not eradicate this species from the UK as it is fully established.

Harvesting or clearance of non-native species has the potential to increase spread where fragments of species that can disperse reproductive propagules or regenerate are broken off and dislodged. Appendix 19 identifies how INNS that are considered likely to cause ecological impacts are spread. Where INNS are present or considered likely to be present understanding the vectors by which these are spread will support control.

Harvesters are likely to avoid heavily fouled seaweeds because of the increase in processing and should be explicitly advised to do so. Removal of fouling species onshore could facilitate spread. Any seaweeds harvested in areas with INNS of concern should be processed away from site. By-catch of INNS should be disposed of in line with a biosecurity plan (Cooke *et al*., 2014) or other codes.

It is unlikely for many species that harvesting of established populations will control the dispersal of populations and consideration should be given as to whether permitting harvesting contradicts messages around control and undermines biosecurity plans by placing a commercial value on invasive species. As noted in the text above, commercialisation and harvesting of seaweeds is unlikely to be supported by the Statutory Nature Conservation Bodies. Attempts to remove invasive algae are often compromised by traits such as asexual reproduction from fragments and regrowth from holdfast attachments if these are not removed completely (McEnnulty *et al*., 2001). Microscopic lifestages that are cryptic and difficult to target and thalli with mature propagules, which may become detached by control techniques contribute to dispersal (Wassmann & Ramus, 1973).

Ideally, if the intention is to permit harvesting to control population growth and spread, species should be targeted before widespread establishment. *U. pinnatifida* was successfully eradicated in New Zealand (Strong, 2003). The population was discovered soon after its introduction and was small (attached to a sunken boat hull). Repeated clearance was successful in preventing this species from establishing. Care was also taken to remove microscopic stages of *U. pinnatifida* and not to fragment the sporophytes, which aid the dispersal of this species (McEnnulty *et al*., 2001).

If harvesting of INNS is supported as a control mechanism, approaches to harvesting should involve stakeholders and consider the ecology of the species to develop effective removal strategies and biosecurity protocols. Consideration must be given to whether the likelihood of control is greater than the risk of spread and dispersal by fragments.

Harvesting should occur prior to fertile periods and be intensive throughout the season to reduce population growth and limit reproductive capacity. All parts of the seaweed should be removed. Removal of biomass before reproduction would reduce the build-up of a seed bank and minimise population increases and spread. Many invasive algal species are able to disperse by drifting and eradication programs that minimise the fragmentation of the collected material are more likely to prevent dispersal.

Harvesting should be intensive with the hand picking of material repeated several times within a year to prevent any individuals becoming reproductive. Unlike native species, harvesting of INNS should remove all the individual including algal holdfasts to inhibit recovery (Davison, 2009).

Removal may also be easier in seasons where individuals are larger and more readily identified. Attempts to remove *S. muticum*, for example, were more challenging in October when light levels were lower and plants were smaller (Critchey *et al*., 1986).

If non-native species are to be harvested, 'Check, Clean, Dry biosecurity principles (see below) should be used when moving between sites to ensure that invasive species, pests and diseases are not spread to new areas.

Management of INNS risks

Management approaches to reduce the risk of INNS spread are discussed in Section 4.3. Management of INNS should consider the risks of spread and how removal of fouling species is undertaken to ensure that these are not released back into the environment. This is particularly key for attached and fouling species which can regenerate from fragments. A number of INNS seaweeds as shown in Table 7 may grow on native seaweeds. Previous guidance documents note that care should be taken when harvesting invasive non-native seaweeds to ensure that seaweeds or spores are not transferred to other areas. Following 'Check, Clean, Dry' biosecurity principles (see Section 4.3), when moving between sites to ensure that invasive species, pests and diseases are not spread to new areas (Bailey & Owen, 2014).

Table 7. Invasive non-native seaweeds that may grow epiphytically on native, harvested species. The priority rank is shown and any evidence that the species is harvested or of commercial interest.

Name	Habitat	Priority	Harvested/Use
Asparagopsis armata	'Attaches to other seaweeds by its barbed branchlets.	2	Ireland: identified as a commercially important species for the production of cosmetics (Sweet 2011a).
Bonnemaisonia hamifera	Grows predominantly epiphytically using hooks to attach.	2	No evidence
Caulacanthus okamurae	Epiphyte and found on artificial structures	1	Grown Korea (Gao *et al*., 2019)
Codium fragile subsp. fragile	Epiphyte	5	*C. fragile*-grown Korea (Hwang *et al*., 2007),

Name	Habitat	Priority	Harvested/Use
			possible recreational harvesting UK
Colpomenia peregrina	Usually epiphytic	5	No evidence
Dasysiphonia japonica	Epiphyte	1	No evidence
Melanothamnus harveyi	Epiphyte	2	No evidence
Undaria pinnatifida	Epiphyte	2	Yes - deliberately introduced to Brittany for commercial exploitation

Invasive non-native species knowledge gaps

All the assessed species were considered likely to occur with harvested seaweeds and associated habitats. Little evidence was found to support assessments of movement by seaweed harvesting. The assessments presented are based on the EICAT assessment methodology and do not present the full range of potential impacts but are focussed on the key pathways resulting in the greatest likely impact on native species and habitats. The chosen EICAT methodology provides a systematic approach for the collation of existing evidence in order to provide a basis for decision making. Nevertheless, in relation to the impacts of INNS the evidence base itself is incomplete. A degree of expert judgement was required in assigning assessments and these should be understood to have a degree of subjectivity and reflect bias in the evidence used and how compilers of evidence sources have reported concerns. A full evidence review and assignation of confidence levels was beyond the scope of this report but even a full review would be subject to limitations around evidence and uncertainty.

The main sources of uncertainty and gaps in knowledge are:

- Lack of detailed ecological evidence for INNS interactions and effects on species, and lifecycle and population dynamics;
- Lack of evidence and understanding on the response of species, communities and habitats to INNS;
- Lack of detailed evidence compared to the EICAT benchmarks and level of effects of INNS; and
- Lack of understanding of the biology, life history and population dynamics of species, the wider indirect links between species, and how those influence the indirect effects of INNS.

Evidence may become outdated as species’ ranges and levels of impacts alter over time. In some cases, in invaded ranges, the number of colonised habitat types is increasing over time and current distribution restrictions may alter as species acclimate to prevailing conditions, or climate change increases habitat favourability.

4. Approaches to management

This section describes general approaches to management and regulation. It does not provide a detailed analysis of seaweed harvesting, protected site management or specific seaweed management and regulation. These are outside the scope of the report.

There is currently no overarching policy or regulation for managing hand collection of seaweed in Wales and England. Harvesting without controls, whether it be by hand or mechanical means, can lead to the overexploitation of the resource, especially if harvesting practices do not allow for regeneration of the seaweed canopy. Statutory, well-structured management plans could help to mitigate the impacts of harvesting and ensure sustainability of the resource, especially when developed in collaboration with scientists, nature conservation authorities and harvesters (Mac Monagail *et al*., 2017). Management plans are supported by ecological understanding of species proposed for harvesting and useful supporting evidence is outlined in Section 2 (above) with further evidence provided in the species appendices (1-14).

Limits on amounts harvested, informed by standing stock assessments, is a key management tool and is discussed further in Section 4.4.

Exclusion of certain sensitive or long lived species has been suggested as a management approach, but is to some extent already applied through other forms of legislation. This is most applicable to Maerl, a calcareous seaweed which can form extensive beds but is extremely slow growing. It has been historically harvested (primarily by dredging but also by hand) for agricultural use, cosmetics and pharmaceuticals.

4.1 Existing regulation in Wales and England

Currently there is no regulatory framework for the management of seaweed resources in Wales and England. Guidance documents recommend all harvesters seek landowner permission, and consult the relevant authorities (the Welsh Government, Inshore Fisheries Conservation Authorities, Natural England or Natural Resource Wales).

Both recreational and commercial intertidal seaweed gathering requires landowner permission from the foreshore or seabed landowners. Commercial gathering on the foreshore or seabed owned by The Crown Estate will also require a licence from The Crown Estate. Before landowners permission or a Crown Estate licence is granted, a Competent Authority under the Conservation of Habitats and Species Regulations 2017 (as amended) may request information from the gatherer to assess whether an adverse effect on site integrity can be ruled out on any Special Areas of Conservation (SACs), Special Protection Areas (SPAs) and Ramsar sites. Natural Resources Wales or Natural England, as the appropriate nature conservation body (ANCB) under these regulations would be consulted and provide advice on the assessment."

Recreational or commercial gathering of seaweed within or affecting a Site of Special Scientific Interest (SSSI) may also require an assent under Section 28H of the Wildlife and Countryside Act 1981 from Natural Resources Wales or Natural England. NRW or NE, as Competent Authority under the Conservation of Habitats and Species Regulations 2017 (as amended) for this assent may request information from the gatherer to assess whether an adverse effect on site integrity can be ruled out on any Special Areas of Conservation (SACs), Special Protection Areas (SPAs) and Ramsar sites.

The level of information that may be requested by The Crown Estate or other foreshore or seabed landowner, NRW or NE, should reflect the scale of the proposed activity. This information could include the following: a biomass stock assessment to predict sustainability of the annual harvest of each species as proportion of the standing stock; a sustainable harvesting management strategy (cutting techniques, rotation, etc.); and/or a monitoring strategy outlining the data and records to be kept.

Reporting of harvested volumes by species can sometimes be a condition of a Crown Estate licence, however, in practice there are no checks in place to ensure this condition is fulfilled.

Tools such as licences with conditions require enforcement, whereas Codes of Conduct outline guidance which may be complied with on a voluntary basis. New national or local legislation would enable enforcement of seaweed harvesting activities, however the introduction and delivery of new legislation is often challenging with limited resources prioritised (see section 4.2 for a summary of existing regulation).

For effective management and planning a greater understanding of the long term effects of hand harvesting on standing stock and sustainability are required, which could be underpinned by records of harvested volumes. For any management to be successful, continual monitoring of both seaweed biomass and the associated marine communities will need to be incorporated, with management plans adapted accordingly in response to changes.

In response to the need for a best practice in harvesting management, Marine Stewardship Council (MSC) and Aquaculture Stewardship Council (ASC) have collaborated to develop a sustainable certification scheme for seaweed harvesting. The MSC-ASC Seaweed Standard became effective in March 2018, and will require monitoring to underpin management (Bennion *et al*., 2019).

4.2 Review of current guidance documents for hand harvesting of seaweed

Details of reviewed Codes of Conduct/guidance documents are presented in Appendix 16. This guidance is evaluated in Appendix 17 which outlines the current guidance measures by species group, with an evaluation of consensus between documents for each management approach, confidence in measure and evaluation of likelihood of compliance by recreational and commercial harvesters. Section 4.3 provides further information on the recommended approaches.

For hand harvesting in the UK, there is little literature on harvesting of small red and green seaweed species. Extensive evidence was found on management of *A. nodosum* harvests from Ireland and Scotland (Hession *et al*., 1998; Guiry and Morrison, 2013; Scottish Government, 2016), and literature developed over several decades of harvesting in the Western Atlantic exists for *C. crispus*, although this should be applied to Wales and England with caution due to differences in growth rate (Mathieson and Burns 1975; Morrissey *et al*., 2001; Pringle and Mathieson 1986). Kelp harvesting is well researched and managed in Norway, although this involves mechanical methods rather than hand harvesting (Vea & Ask 2011).

Approaches to mananagement from Codes of Conduct and identified in the evidence review are listed below, they are not recommended or applicable for every seaweed group.

- Harvesting methods
  - Cutting height, leave a proportion of the plant (holdfast and some frond) remaining at the base;
  - Selectively cut with scissors rather than plucking or uprooting to support recovery and reduce by-catch;
  - Avoid by-catch of epiphytes and vulnerable species
  - Avoid harvesting reproductive material if possible (or only take half from each plant in the case of H. elongata);
  - For certain species (e.g. *for F. serratus* and *F. vesiculosus*), only harvest part of mature plants
- Harvesting period
  - Harvest during active growing season;
  - Avoid harvesting during the reproductive season;
- Harvesting frequency
  - Fallow periods for recovery of canopy in A. nodosum and perennial kelps
- Harvest limits
  - Quotas, volumes or bag limits;
  - Proportion of standing stock biomass removed/left remaining
- Harvesting spatial considerations
  - Harvest sparsely, leaving unharvested plants between those taken;
  - Shape, size and spacing of harvested areas (i.e. between harvested plants or patches);

Not specifically covered in guidance but considered useful are:

- Biosecurity planning for invasive non-native species and
- Community management and co-governance.
- No-take protected areas, including those designated by existing legislation (as a reference area, to protect a source population, or maintain ecosystem services such as biodiversity or coastal protection);
- Spatial, temporal and seasonal closures (e.g. in response to stock declines, during peak seaweed reproductive times, seasons of slow growth, or during breeding/nursery periods for associated marine life)

4.3 Review of management approaches

Harvesting methods: Cutting height (distance above the base)

Generally (as for example with *A. nodosum*), the more of the plant left by harvesting, the more quickly the biomass will recover. The effectiveness of cutting height approaches are summarised for each species group in Appendix 17 as they form a key part of Welsh and English Codes of Conduct (Bailey & Owen, 2014; NRW 2018). Any species-specific information found regarding cutting height recommendations is provided in the species dossiers Appendices 1-14.

For many species, the advice is simply to leave the lower part of the frond and holdfast. Kelps grow from the meristem, so require that the basal portion of the blade is left to allow regrowth. Advised *A. nodosum* cutting heights are variable, with heights of 10-20 cm recommended for Wales and England (NRW, 2018; Bailey & Owen, 2014); 25 cm for Ireland (Bruton *et al*, 2009) and 30cm for Scotland (Scottish Government, 2016).

Regulations in Maine, USA, stipulate a height of 40.6 cm, which along with other measures appears to be more effective in minimising impacts on mobile megafauna (crabs) associated with the weed (Phillippi *et al*., 2014).

Harvesting methods: Selective harvesting to avoid by-catch

Selective cutting methods used by hand harvesters can avoid removing holdfasts or basal crusts (to support regeneration) are likely to have less bycatch than mechanical methods, however eggs, epiphytes and epifauna and sessile or slow moving species including protected species like stalked jelly fish may be removed with seaweeds. By-catch of rare, protected species should be avoided.

Best practice recommendations are to select fronds with minimal epiphyte growth and attached animals and to remove and return obvious species (but not invasive non-natives, see Section 3.4).

Harvesting methods: avoid reproductive material

It is recommended that harvesters avoid removing reproductive material. The presence and location of this will vary throughout the year by species as outlined. Further information on reproduction is provided in Appendices 1-14.

In Laminarian kelps reproductive ‘sorus’ tissue forms on the blade, whereas in *A. esculenta* and *S. polyschides,* sorus material occurs within reproductive structures, called sporophylls, found at the base of the stipe. As such, if hand harvesting targets the blade and takes place during the reproductive season then in Laminarian kelp the opportunity for reproduction will be lost until regrowth takes place (Burrows *et al*., 2018), whereas *A. esculenta* and *S. polyschides* would still be able to reproduce. The location of spore release (i.e. higher or lower in the water column) also has implications for dispersal, with potentially greater distances likely if released higher above the seabed than lower, where water flow may be dampened by the seaweed canopy.

In both *Fucus* spp. and *H. elongata*, fertile material is likely to be removed by harvesting: reproductive receptacles on *Fucus* form at the tip of the frond, while harvesting of *H. elongata* targets only the reproductive structures. As such, guidance (Bailey & Owen, 2014; NRW 2018) suggests removing only half of the *H. elongata* straps from an area (see section 4.3, Appendix 16, and species dossier Appendix 6 for more detail). The number of reproductive receptacles in *Fucus* increases with age. As large individuals contribute disproportionately to reproduction, removal of these individuals could potentially impact reproductive output of the population.

C. crispus and *M. stellatus* produce fruiting bodies across most of the upper part of the frond, which could be removed if harvesting occurs during peak fertility (autumn and winter for *C. crispus*).

For opportunistic species more of the blade supports reproduction. In *Porphyra* spp. the reproductive bodies are spread across most of the blade. In *Ulva* spp., every cell across the blade can be converted into spore production, generating a huge reproductive output. For this species, selective cutting may be less critical.

Harvesting periods

The theory underpinning advice in the literature with regard to seasonal closures to protect the stock during periods of peak growth is contradictory. Lotze *et al*, (2019) recommend closures (of mechanical harvesting) during peak growing periods, in order to protect growth. However Codes of Conduct (e.g. Bailey& Owen 2014) generally advise that harvesting activities should take place while growth is most rapid, to allow for faster recovery. It is the opinion of this report that, for a management goal of rapid recovery from hand harvesting, activities should take place during peak growth (in accordance with Bailey& Owen, 2014). There is general agreement that harvesting of reproductive material should be avoided if possible, whether by selective cutting or seasonal closure.

McLaughlin *et al*. (2006) report that when stocks of *Porphyra* spp. are very low, harvesters travel to Western Scotland to supplement the supply. While driven by an already depleted resource, their response could potentially act as a de facto closure. If better monitoring were in place, closure or limits to harvesting could be applied at intensely targeted sites (such as Freshwater West), although would require enforcement resources and education.

Area-based closures to mechanical harvesting of *L. hyperborea* are implemented in Norway if/when required (e.g. if urchin grazing becomes a threat, or if kelp standing stock is deemed too low). There is a strong monitoring program in place to inform management response.

In southern Norway, summer die-off events have been recorded for both the sugar kelp *S. latissima* and *L. hyperborea,* associated with high-temperature events (Vea & Ask 2011). Management structures allow for the closure of the Norwegian kelp fishery in areas where warm-water events have occurred (Vea & Ask 2011).

Harvesting frequency and fallow periods

Sufficient time should be allowed between harvests for regrowth of the standing stock biomass and the associated ecosystem. This will vary based on the recovery rate of the species and site specific environmental conditions and favourability. For example, in Ireland it is recommended that *C. crispus* can be harvested every 2 years at exposed sites but every 4 years at sheltered sites (Edwards *et al*., 2012). For the majority of species harvested by the Cornish Seaweed Co. (see Appendix 22) sites are harvested only once per year. Exceptions are for fast growing *P. palmata* and *Ulva* spp. which recover quickly enough to allow for harvesting at most sites twice a year.

Fallow periods may be voluntary or statutory, and if followed have been demonstrated as effective in mitigating impacts by allowing recovery, however rotation of cropping requires management, be that by an individual harvester or by a regulator. Application of fallow periods are summarised for each species group in Appendix 16.

Mechanical harvesting fallow periods for kelp are covered in a sister report (Wilding *et al.,* in prep). Kelp canopy regeneration following mechanical harvesting requires 4-5 year fallow periods. No evidence was found for this measure applied to hand harvesting of kelps.

Harvest limits (quotas, volumes or bag limits)

Limits on the amount (biomass) of seaweed that can be removed is a key management tool, examples found by this report are summarised in Table 8 (below) and provided in Appendix 16. Limited literature was found on harvested quantities in Wales and England, as no official figures are reported and commercial harvesting businesses are reluctant to disclose this information. Recreational harvesting is similarly difficult to record because it usually goes unreported (although is the subject of a PhD thesis currently in preparation – Morris-Webb pers. com). Sections 4.4 and 4.6 discuss further considerations around biomass estimation and monitoring.

Quotas may be more appropriate for commercial scale harvesting, while bag limits may be applicable to personal or artisanal scales. Disadvantages of bag limits are that they will be difficult to enforce and may create an increase in collection (McLaughlin *et al*., 2006).

For the management of commercially harvested *A. nodosum* in New Brunswick, Canada, estimates of the standing stock and productivity were used to establish annual quotas. Following a precautionary principle, 7% of the stock can be harvested annually. Harvesting applications must submit a detailed annual harvesting strategy outlining proposed harvesting zones, rotation frequency, monitoring and controls, as well as submitting harvesting data (Sharp & Bodiguel 2001; Ugarte & Sharp 2001).

The recommendation (see Appendix 16) to remove no more than 1/3 of the plants from a shore in a given year (Angus, 2017), and to collect less than 1/3 of each individual plant (Bailey and Owen, 2014), appear to be arbitrary, but offers a sensible and easy to follow standard based on the precautionary principle. Applying this for sites where there are multiple harvesters operating could be more difficult and recreational hand harvesting based on this guidance could see progressive denudation at popular sites, where multiple harvesters each remove a proportion of the stock.

Table 8. Current management guidance that considers biomass.

Reference	**Region**	**Species**	**Sustainable harvesting advice**
Scottish Gov, 2016	Scotland	Various species: Includes regeneration periods	Prior to harvesting undertake assessment of stock and biomass, estimate % cover
Morrison, 2018	Scotland	Kelps	Of 20Mt *L. hyperborea* - 6.5 Mt predicted to be in harvestable area where biomass exceeds 5 kg/m^2
O'Hanlon & O'Hanlon, 2018	Ireland	*F. serratus,* *F. vesiculosus*	Standing biomass of the 2 target seaweeds is 76.3 tones Seaweed gathered in 40kg sacks and 1-2 sacks at low tide period Annual harvest quantity = 2,000kg (2 tones wet weight, 50 sacks over the year, ~1 sack a week).
Government Jersey, 2019	Jersey	Various	Harvesting of attached seaweed will only be permitted by non-mechanized means Daily bag limits for the majority of red and brown seaweeds be set at 5kg for recreational and 10kg for commercial
Burrows *et al.,* 2018	Scotland	Kelp	Proportion of standing stock advised for harvest varies by site from 10-35%, e.g. 6.5 Mt of *L. hyperborea*
Burrows *et al.*, 2010	Scotland	*A. nodosum*	Outer Hebrides was estimated at 170,500 tons of which 15,000-25,000 tones could be sustainably harvested annually

Harvesting spatial considerations

Consideration of the shape, size and spacing of harvested areas (i.e. between harvested plants or patches) can reduce impacts and support recovery. Within any harvested area it is recommended to leave an unharvested area as a source population to facilitate recolonisation (McLaughlin *et al*., 2006). The effectiveness of these approaches are summarised for each species group in Appendix 17.

Some recommendations were found in the literature for spacing (see Appendices 1-14) For example, Edwards *et al*. (2012) recommend that mature *C. crispus* plants should be left every 2m. UK Codes of Conduct recommend that unharvested *A. nodosum* should be left between harvested plants to maintain habitat value end ecosystem function (NRW 2018; Bailey& Owen 2014). In Quebec, 50% of any given area must be left unharvested (Grendon *et al*., 2017). Leaving some canopy in place through managing cutting height and leaving unharvested areas will support recovery (see Appendix 2). Similarly, leaving some of the turf of red algae may prevent the shifts in community composition observed following removal of *C. crispus* (see Appendix 3) with shifts to *Fucus* spp. (Lubchenco 1980) and *Corallina officinalis* and encrusting coralline biotope observed (MacFarlane, 1952).

Harvesting of *Porphyra* spp. in South Africa was recommended to be managed by limits on frequency and harvesting from patches, regularly spaced apart, with dense patches left throughout the shore to allow for growth and reproduction (Griffin *et al*., 1999). This approach is considered not only to sustain the resources but also to mitigate impacts on associated fauna, such as intertidal molluscs.

Spacing of mechanical harvesting for kelp and *A. nodosum* is covered in a sister report (Wilding *et al.,* in prep.).

No-Take Zones

Harvesting No-Take Zones are recommended to protect designated sites, vulnerable or sensitive features (e.g. maerl habitats, above, or features vulnerable to trampling / erosion / disturbance from harvester access), maintain source populations, promote conservation, maintain ecosystem services and as a reference / control area against which harvesting monitoring can be evaluated. They may be voluntary or statutory and should be assigned adjacent to the harvesting area. They could be permanent exclusion zones (which are simpler to administer and can offer greater protection) (McLaughlin *et al*., 2006) or seasonal closures (as above).

Community management and co-governance

No evidence was found for this approach within the time frame of the review.

Biosecurity plans for invasive non-native species

Harvesting of native species should also consider the current presence and status of INNS. The development of biosecurity plans to manage INNS risks would represent best practice and is recommended. This would be particularly relevant if a harvesting company was harvesting across several sites in an area and moving equipment between sites Appendix 23 outlines how a company manage risks by only visiting one site and checking, cleaning and drying equipment before another site visit.

Harvesters should follow, ‘check, clean, dry protocols to limit the spread of INNS. Equipment and clothes should be checked. Vehicles may also transport terrestrial INNS and where necessary should be cleaned.

Check your gear after leaving the water for mud, aquatic animals or plant material. Remove anything you find and leave it at the site.

Clean everything thoroughly as soon as you can, paying attention to nets, waders, and areas that are damp and hard to access. Use hot water if possible.

Dry everything for as long as possible before using elsewhere as some invasive plants and animals can survive for two weeks in damp conditions.

Where INNS are present, heavily fouled seaweeds should not be removed, any processing of seaweeds with some INNS attached should take place away from the site with INNS disposed of rather than returned to the shore.

Statutory closures

No literature was found for statutory closures to hand harvesting within the scope of this review. For example in response to seaweed biomass declines, during peak seaweed reproductive times, seasons of slow growth, or during breeding/nursery periods of associated marine life.

4.4 Biomass stock assessments

Assessment needs and methods

In order to appropriately manage seaweed harvesting, the distribution and standing stock biomass of the resource must be estimated. This can be difficult for species with highly variable or seasonal growth, such as *Porphyra* spp. (Bunker, 2021), and will vary greatly between sites. There is an absence of evidence for standing stock biomass for most species in the UK. Where evidence does exist, it is usually spatially restricted.

Here we assess some simple methods of assessing this resource in the UK for management purposes and make some initial estimates of biomass of intertidal seaweed in an area encompassing Wales and part of the Bristol Channel.

There are currently no comprehensive figures of annual seaweed production for the UK, which could be submitted to the FAO or used to inform management. Neither are there clear estimates of the seaweed standing stock biomass available as a potential resource for harvest (Capuzzo & McKie 2016).

Assessing stock biomass

Estimation of the biomass of seaweeds available for harvesting requires understanding of the extent of habitats suitable for such seaweeds and the typical abundance of each species in these habitats. In similar previous studies, a typical approach would be first to commission biomass density surveys to give estimates of the biomass per unit area as wet weight of plants per unit area (kg/m2), followed by estimation of the extent of the supporting habitat. Where direct mapping of the seaweed habitat is not possible, statistical models can predict this extent, using measures of the association of seaweed abundance with environmental factors like wave exposure, temperature, or aspects of water quality (see also Appendix 22 and Section 4.6 below).

Size of projected harvests requiring a biomass survey

The main target species for mechanical harvesting in the North Atlantic subtidal are the large kelps *L. hyperborea*, harvested typically in Norway using a rake-type dredge, and *L. digitata* in France using a rotating grapple or ‘scoubidou’ (Burrows *et al.*, 2018, Mac Monagail *et al.*, 2017 for a recent reviews). The extent and magnitude of these mechanical harvests (1000s to 10000s tonnes per year) have long demanded good estimates of the stocks of these species.

In the North Atlantic intertidal, the main target species for harvesting in terms of biomass is *A. nodosum*. In the Outer Hebrides, a decision was made to estimate the biomass of the species in the harvest area once the local harvest had grown to a level that was perceived to be an appreciable proportion of the annually renewed fraction of the harvestable stock. The decision to commission detailed surveys does presuppose that a broad idea of available stocks already exists, which in this case was based on earlier intertidal surveys from the 1940s and 1950s (Walker, 1947). While the whole archipelago was estimated to support 170kt of wet biomass, only 60kt was within a distance from landing points that allowed economically viable transport of the harvest, and only 25% or 15kt of that 60kt was thought to be renewed annually through regrowth after harvesting (Burrows *et al.*, 2010). Thus, the maximum sustainable harvest emerged as potentially about 10% of the total stock.

McLaughlin *et al*. (2006 and references therein) detail biomass and standing stock estimates for *A. nodosum*, kelps and fucoids in Northern Ireland. They define three harvesting scales based on both the volume of each species harvested and the proportion of the standing stock biomass taken over a given area, inclusive of both hand and mechanized methods (McLaughlin *et al.*, 2006). These are small <1kg; artisanal 10’s of kg per species per day, not collecting more than 25% of the biomass per 100 m^2 and not reharvesting until re-growth has occurred; and commercial as 110’s-1000’s tonnes per business, more than 5% of the standing stock per km of coast (McLaughlin *et al.*, 2006).

As a precaution therefore, it would seem prudent to commission detailed biomass surveys for species when the planned level of harvesting approaches an appreciable fraction of estimated annually renewed biomass for that species in the harvesting management area. What that fraction might be would be a choice for managers based on their level of concern given an appreciation of the risks of lasting damage to stock levels (see recovery rates, Section 2.2). A recent study on populations of *A. nodosum* at its most southerly distribution limit in Portugal provides supporting evidence for the establishment of management plans for “edge populations” (Borges *et al.*, 2020). For *A. nodosum*, biomass renewal was estimated to be only 0.94% of the standing stock, which in combination with the vulnerability and ecological relevance of range edge populations suggested wild harvesting was unsuitable.

Below we present guide levels for intertidal seaweed stocks based on wet biomass, with the caveat that for some species little is known of their ability to recover after intense harvesting. While carefully cut, hand-harvested *A. nodosum* with basal meristem tissue can recover within 5 years and dulse *P. palmata* and laver *P. linearis* may recover within a year (Stagnol *et al*., 2013).

Biomass assessment conclusions

The total biomass of intertidal macroalgae in each region depends on the extent and area of habitat that potentially supports such growth. Appendix 23 provides an example of biomass estimation of biomass for Wales and England and Scotland. This approach shows how quantitative survey information combined with environmental data can be used to make useful and potentially robust estimates of the biomass of intertidal seaweed available for harvesting. While we have not developed the habitat suitability models used in similar studies, this exercise demonstrates the value of the broad approach. Biomass density estimates from Scottish Association of Marine Science (SAMS) survey data collected in Scotland may give a first approximation of the expected biomass among species in Wales but would ideally be supplemented or replaced by locally collected data, especially for the species likely to be hand harvested.

Total biomass values are obviously not direct indications of what quantities can be sustainably harvested. Such limits need to take account of the ability of species to recover and the area available for harvesting (Burrows *et al*. 2010). Much of the exposed rocky coastline of Wales will be inaccessible to harvesters, reducing the total biomass available and potentially focussing the harvest on a much smaller area with greater impact. Elsewhere, such as in Dorset and south Devon where there is a lot of interest in commercial harvesting, SNCB (Statutory Nature Conservation Bodies) advice is to avoid designated sites where possible. There are very few areas outside of MPAs along this stretch of coast, and these may be intensively harvested by multiple companies due to lack of availability of harvest sites in this MPA-packed area. GIS methods are likely to be able to identify accessible areas and to reduce totals accordingly.

These figures may be used to judge the need for stock surveys given the size of any proposed harvest and, given that they are spatially referenced, could be subsetted to give the estimated biomass in a particular planned area of harvest.

4.5 Management technique evaluation

A number of management tools are available and approaches should be tailored to meet the individual species, region and proposed harvesting regime in question. These include seasonal closures, mandated fallow periods, closed areas, selective and partial harvesting, and total allowable harvest (reviewed regularly). As these techniques are rarely applied separately, it is difficult to assess their specific respective success in terms of nature conservation.

Appendix 17 presents a tabulated evaluation of confidence in current management guidance for Wales and England, based on the consistency between management documents, evidence base for measures, and likelihood of uptake by harvesters.

There is a broad agreement that small scale (i.e. artisanal not industrial) hand harvesting is unlikely to result in substantial adverse environmental impacts although small green and red seaweeds could be completely removed by these methods (Scottish Government, 2016). Examples of over harvesting by hand methods include *H. elongata* in France and Portugal, see Mac Monagail *et al*., 2017). There are concerns about the risk of smaller species (*C. crispus, M. stellatus*, *Ulva* spp. and *Porphyra* spp) being entirely removed from large areas of shore in Wales and England by intensive or cumulative harvesting activity (Perry *et al*., 2014) and in Scotland (Scottish Government, 2016).

To avoid overharvest for recreational use at popular sites, management strategies must include capacity to monitor the cumulative effects, for example by creation of a public register of harvesting activities as part of any future regulatory process (Scottish Government 2016; Perry *et al*., 2014).

An effective management strategy will utilise a combination of management approaches. For example, sustainable harvesting of *A. nodosum* appears to be possible, and has been the subject of extensive research. Management includes cutting at a certain height (usually 20-25cm), and rotating harvested areas with fallow periods of 3-5 years (Baardseth, 1955; McErlean *et al*., 2002; Guiry & Morrison, 2013), although recovery rates vary widely by region, with between 11 months and four years reported in Ireland (Boaden & Dring, 1980; Kelly *et al*., 2001).

Norway has developed a model for management of *L. hyperborea* harvest (via mechanical trawl), with fallow periods (4-5 year rotation), seasonal and spatial restrictions for harvesting, and allowance for area-based closures if/when required (e.g. if urchin grazing is a threat, or if kelp standing stock is deemed too low). Measures are supported by a well-resourced plan for research and monitoring to assess the wider impacts of (Meland & Rebours, 2012; Vea & Ask, 2011). The approach has generally been seen as favourable, with the seaweed resource appearing to be stable, although some concerns regarding the longer-term environmental impacts have been raised. However, the kelp resources of the UK are considerably smaller. Large scale kelp harvesting, for industrial scale applications such as alginate extraction, is potentially a future concern for Wales (Perry *et al*. 2014), however as the volumes concerned are likely to require mechanical methods or cultivation, this is considered beyond the scope of the current review.

Self-regulation by harvesters

Statutory regulation for seaweed harvesting is absent from many European countries. Even in Norway where kelp stocks are considered to be relatively sustainably managed by national legislation, mechanical harvest of foreshore algae such as *A. nodosum* is regulated by only private owner rights. There are no statutory ‘fallow periods’ in place for harvesting *L. digitata* in Brittany, with most beds harvested annually, however some local harvesters self-managing their fishery, leaving 1-2 years before re-harvesting (Werner & Kraan 2004).

Self-management strategies appear to be effective when access to a resource is exclusive, generating high motivation for sustainable management. For example, the Cornish Seaweed Co. have relatively little competition for the seaweed resource they target, and operate with a strong sustainability and monitoring strategy (see case study in Appendix 23).

In Scotland, commercial hand-harvesting of *A. nodosum* takes place in the Outer Hebrides by two companies; The Hebridean Seaweed Company and Uist Asco, which operate in different areas, using a combination of boat and rake, mechanical vessels, and hand harvesting methods. Harvesters from both companies cut the seaweed 12 inches (30 cm) from the base, and then leave the bed to regenerate for 3-4 years before returning, with individual cutters managing their own cutting areas. This combination of management approaches, together with the motivation generated by self-management, appear to support a sustainable industry. For more detail see Exclusive access vs cumulative effects below and Appendix 2 for specific information on *A. nodosum*

Exclusive access vs cumulative effects

The measures discussed above are suitable for use by individual business, or incorporation into regional or national seaweed management plans. In situations when one business or harvester has exclusive access to a seaweed resource there is high motivation to maintain the stock sustainably. However, when multiple harvesters utilise the same areas, cumulative effects are likely to be an issue. For example, the CSC have a licence to harvest over an area of coastline, and are the only commercial operators in the area. While some recreational harvesting is likely to take place, none was reported in the literature, hence it is likely to be of limited intensity. The CSC operate in line with national harvesting codes of practice, managing rotation between different areas to allow regrowth. Similarly, one company has commercially harvested *Porphyra* spp. at Freshwater West in Wales for many years and appears to manage the resource sustainably (Bunker 2021). However, Perry *et al*. (2014) raised concern about *Porphyra* spp. stocks at Freshwater West due to intensive cumulative recreational harvesting, which will increase pressure.

Cumulative effects may be of concern where areas available to harvest are limited. For example, pressures on available sites could increase where there is a lot of interest in commercial harvesting and very few areas outside of Marine Protected Areas.

Over harvesting by hand methods can still occur (for example *H. elongata* in France and Portugal (Mac Monagail *et al*., 2017) and is intuitively more likely when multiple harvesters are utilising the same resource. There are concerns that small, relatively slow growing reds (*C. crispus* and *M. stellatus*) and even faster growing *Porphyra* spp. and *Ulva spp.* could be completely cleared from an area by repeated or cumulative hand harvesting in the UK (Scottish Government 2016; Perry *et al*., 2014). Management strategies must include capacity to monitor the cumulative effects of multiple harvesters.

4.6 Monitoring approaches to support management

Monitoring is a key component of management approaches to seaweed harvesting, as effective monitoring will capture changes in population structure through time, allow for natural and anthropogenic pressures to be disentangled, provide opportunities to alter management approaches to achieve sustainability. The specific approach to monitoring will be dependent on a number of factors, including proposed harvesting intensity, characteristics of the site and wider region, targeted species, management objectives and available resources.

A variety of survey methods are available when designing a monitoring programme, which will again depend on the species and region in question, its distribution, and available resources (see also Section 4.6 above on biomass estimation). Traditional techniques involving shore-based surveys (along transects, grids or haphazardly as appropriate) can be used to rapidly obtain valuable data on species' abundance, biomass and distribution. Surveys should be conducted by experienced surveyors with adequate training in taxonomy and ecological survey techniques. For some of the larger, more conspicuous and widespread species, remotely-sensed imagery (i.e. from satellites, aerial surveys or drones) can be used to benchmark seaweed populations and monitor changes (Brodie *et al*., 2018). Recent advances in automated image classification can be applied to seaweed populations to obtain rapid and reliable estimates of cover and biomass (Mahmood *et al*., 2020). In tandem, species distribution models can be developed to predict, given environmental conditions at a certain site, the likely abundance and biomass of targeted species, as has been developed for *L. hyperborea* in Scotland (Burrows *et al*., 2018). Such models can be used to extrapolate from survey sites to the wider region, and to identify sites or regions where seaweed populations are less abundant and more restricted than environmental conditions would predict, thereby indicating impacts of harvesting or other local pressures.

Methods for assessing stock biomass

If the overriding objective is to achieve sustainability, a robust and extensive survey of standing stock and aerial extent of targeted species should be conducted prior to establishment of harvesting to establish a reliable baseline against which to detect change. However, as seaweed populations can be spatio-temporally highly variable, such baseline surveys need to be designed and conducted to adequately capture variability patterns. For some species in some regions, particularly those with patchy or restricted distributions, this may require significant sampling effort and become resource intensive. For regions and species with homogenous and widespread distributions and high abundances, sampling effort and resource requirements may be relatively limited. In all cases, following an initial survey, population-level data should be explored (e.g. with power analysis) to determine what level of sampling would be required in the future to detect different thresholds of change (i.e. 10, 20, 50% loss). Data should be also be explored to offer guidance on the sampling frequency required to detect such changes. Despite the necessary effort the importance of monitoring has been recently demonstrated for *A. nodosum* in Milford Haven, where fixed quadrats have demonstrated declines resulting in recommendations not to harvest (Bunker *et al*., 2021).

For kelps in the shallow subtidal, such as *L. hyperborea*, estimates of the population density of plants can be made directly by divers making counts and measurements of plants in sampling quadrats (Smale *et al*., 2020, Kain, 1962) and by observations from the surface using direct observations from small boats (Bekkby *et al*., 2019). Early efforts to assess these kelps used modified spring grabs (Walker& Richardson, 1955) to bring up samples from the seabed. This method allowed spatially extensive assessments of the presence and biomass per unit area of the species and ultimately a stock estimate for harvestable *L. hyperborea* in Scotland at 10 million tonnes wet mass. While this method certainly underestimated the density and biomass of plants because of the less-than-100% efficiency of the spring grab method, the 1940s and 50s scheme remains the most successful effort to survey coastal seaweeds for harvestable biomass in the UK to date. More recent methods have used small boats that can operate in shallow water kelp habitats, combining acoustic detection of kelps as above-bed backscatter with drop-down cameras for species and density assessment (Blight *et al*., 2011).

For intertidal macroalgae, complete access at low tide allows size and density of plants to be directly measured. Transect methods using sampling quadrats allow precise quantification of the amounts of each species available at every level between low water and the upper biological limit of macroalgae. For *A. nodosum* in the Outer Hebrides for example, a level and measuring pole were used to establish the shore profile and the vertical tidal levels of the upper and lower limits of the species at each survey site. All plants were removed, weighed and measured in multiple stratified 0.25m2 quadrats in the seaweed zone (Burrows *et al*., 2010). Variations on this method have been used successfully for decades (Mann, 1972).

Remote sensing using unmanned aerial vehicles (UAVs), aircraft and satellites offers the possibility of estimation of the extent of intertidal seaweed beds (Tait *et al*., 2019), but poses significant challenges for subtidal seaweeds given the lack of visibility below depths depending on the light attenuation of the local waters (Uhl *et al*., 2016, Casal *et al*., 2011). High resolution multispectral satellite images, such as Sentinel 2, can resolve intertidal seaweed belts and can be effectively used to give estimates of the quantity of bedrock covered by macroalgae, and help the estimates of total seaweed stocks in a region (Lewis, 2020). Current JNCC monitoring guidelines for habitat mapping using UAVs (Crabb *et al*., 2019) are strong on operational planning and execution but do not give methodology specifically targeted at assessing the extent of intertidal seaweed beds, beyond direct visual interpretation of the product.

Methods for continued monitoring of harvested sites

Once a significant harvest has begun, it may be necessary to continue monitoring the state of the exploited stock. Given that it may be in the interests of the harvesters that their stock is not diminished or unduly damaged, and that they may be the ones most frequently present in the harvesting area, continued monitoring in some form could be a condition of any license to exploit wild stocks. Stock assessments by harvesters would probably work best where areas are licensed to a single harvester or company and thereby a long-term interest in maintaining the stock. The extent and frequency of continued monitoring would depend on the nature and species involved in the harvest. For short-lived species in wave exposed or otherwise highly disturbed habitats, it would be important for the monitoring to distinguish the effects of natural seasonal variations in biomass and those due to extreme event such as storms, from the effects of the harvesting activity itself. To achieve this, ideally a monitoring scheme would include assessments of seaweed stocks in harvested and non-harvested areas (Stagnol *et al*., 2013). A full Before-After/Control-Impact (BACI) survey design (Underwood, 1994) would give the right level of confidence in the effects of the harvest on the stocks, although may not be feasible in all cases due to cost.

Biomass management issues may occur where there is unregulated and unquantified recreational harvesting carried out alongside commercial harvesting.

5. Report summary, conclusions and recommendations

5.1 Key harvested species, growth and recovery

Commercial operators are likely to target selected, high demand species with up to 20 species known to be harvested commercially in Wales and England (Morris-Webb in prep 2021). Commercially in Wales and England, Sea spaghetti *H. elongata*, dulse, *Ulva* spp., *Porphyra* spp., *C. crispus, F. serratus* (some *F. vesiculosus*) and the kelps *L. digitata* and *S. latissima* are the key species targeted.

Recreational harvesting activity targets 23 seaweed species in Wales. Of these, Laver, *Porphyra* spp. is of particular importance to recreational harvesters in South Wales, followed by bladder wrack (*F. vesiculosus*), pepper dulse (*Osmundia* spp), kelps, carrageen (*C. crispus* and *M. stellatus*) and dulse (*P. palmata*) to be of particular importance.

The majority of hand harvesting activity is for food use, with limited amount taken for cosmetics and pharmaceuticals. A small amount of mature adult plants are also hand harvested to provide fertile material for seaweed cultivation (see Wilding *et al*., 2021 in prep, for further detail). Little evidence for collection of beach cast seaweed was found in Wales and England, which is thought to be opportunistic and seasonal following winter storms, with applications as fertilizer or soil conditioner.

There is considerable information available for certain aspects of some harvested species such as distribution, growth and seasonality and ecological importance. This report presents key information, illustrated by examples, with more detailed evidence compiled in the species appendices (Appendices 1-14). Detailed reviews for *F. spiralis*, *P. canaliculata*, Corallina species, *Gracilaria* spp. and *Osmundea* spp. have not been included due to lack of commercial harvesting and limited evidence.

Seaweed species are distributed throughout Wales and England but most are absent from the South East coasts of England due to a lack of suitable rocky substrata. In the more wave-exposed areas which characterise the south and west, brown seaweeds and small perennial red seaweeds dominate the shore, while kelps *L. hyperborea* and *L. digitata* are found on the subtidal fringe. The kelp *A. esculenta* is found at only the most exposed sites, while *A. nodosum* requires shelter. Unusually, Laver *Porphyra* spp. thrives in situations of extreme sand scour. Species at their geographic range edge are less resilient to harvesting pressure, due to the additional stress of sub-optimal environmental situation. Cold adapted kelps *A. esculenta* and *L. digitata* are declining towards their southern range edges in England and Brittany respectively, while the short lived opportunistic kelp *S. polyschides* may be increasing in abundance, exerting competitive pressure on other kelps.

Hand harvesting methods usually involve selective cutting of the frond with scissors or a small knife, although some harvesters (presumably more recreational than commercial) are thought to pluck seaweeds by hand. Access by foot from the shore is most common, with a small number of commercial operators using diving or free diving methods for species found lower on the shore (e.g. *H. elongata* and kelps).

Growth rate, life history and reproduction which underpin recovery from harvesting are species specific and spatio-temporally variable. *A. nodosum, L. hyperborea* and *C. crispus* are well evidenced, whereas fundamental details of some key species are lacking, or just emerging (E.g. *Porphyra* spp., Knoop *et al.,*2020). *A. nodosum* is the most slow growing and long lived of harvested species, followed by the kelp *L. hyperborea.* Perennial red seaweeds *C. crispus* and *M. stellatus* are also relatively long lived, followed by other kelp and wrack species. By contrast, *S. polyschides Ulva* spp. and *Porphyra* spp. are fast growing and short lived.

Growth from kelps take place from the meristem and the base of the blade, while *A. nodosum* and *Fucus* spp. grow from the apical tips. Kelps will die if the stipe is cut, whereas many species (*Fucus* spp. *P. palmata, C. crispus* and *M. stellatus*) can regenerate vegetatively from the holdfast (and basal crust in red seaweeds). Regrowth is more rapid if fronds are cut higher above the holdfast, particularly in slow growing *A. nodosum* which will regenerate in 3-5 years following cutting at a height of 20-30cm. Most species can regenerate from partial cutting of the blade, with *F. vesiculosus* and *P. palmata* growing prolifically from damaged frond edges. *Ulva* spp. and *Porphyra* spp. also regrow rapidly.

Peak growing and reproductive seasons vary greatly between species. Reproductive material in *A. esculenta* and *S. polyschides* is found on specialised structures called sporophylls located near the base of the stipe which are easy for harvesters to avoid. For all other species, selective cutting is likely to target reproductive material when it is present. This is particularly the case for *H. elongata*, whereby 98% of the biomass is reproductive tissue.

Dispersal potential of most species is poorly understood but is thought to be low (10’s to 100’s m from the parent plant), with the exception of *Ulva* spp. which can rapidly colonise new areas up to 35km from source populations. If overharvesting clears adult seaweeds from an area, recovery may occur from less visible life stages, such as the microscopic gametophytes “seed bank” in kelps or the basal crust in perennial reds (*P. palmata, C. crispus* and *M. stellatus*). Recruitment is promoted by the presence of adults, which provide protection from desiccation and irradiance, but also limit light reaching juveniles. Success of recruitment is highly variable and appears to be lowest for *A. nodosum* which experiences 99.9% mortality within the first year. *A. nodosum* populations are therefore maintained by vegetative growth of basal shoots.

For hand harvesting in the UK, there is little literature on harvesting of small red and green seaweed species. Extensive evidence was found on management of *A. nodosum* harvests from Ireland and Scotland for *C. crispus* from the Western Atlantic. Kelp harvesting is well researched and managed in Norway, although this involves mechanical methods.

5.2 Harvesting Impacts

Any harvesting activity will have some environmental impacts, and even with the most robust seaweed management plan in place, there may still be wider impacts to marine ecosystems such as a reduction in habitat available for other species (e.g. invertebrates and fish) or reduction in detrital material entering coastal food webs.

The reduction in canopy biomass of large target species is relatively easy to manipulate and study, hence the relatively good evidence base for key species which have historically been of commercial interest. Phase shifts or short-medium term abundance changes from one dominant canopy forming species to another are relatively well documented in response to hand harvesting of species including *A. nodosum* (Tyler 1994) and *C. crispus* (Sharp *et al*., 2006).

However changes in the composition of associated communities and specific chemical or physical impacts are less well documented, with variable coverage in the literature. For example, while kelp canopies are known to provide protection from coastal erosion by dampen wave energy, only one study was found which investigated the impact of canopy loss on delivery of this function.

Impacts on mobile species and nursery habitat function are difficult to quantify, as it is typically difficult to discern whether the absence of these species is the result of harvesting-induced mortality or simply emigration into adjacent habitats. One exception is Kelly *et al*., (2001), who recorded that a decrease in periwinkles within harvested *A. nodosum* patches over winter was mirrored by a corresponding increase in adjacent control sites, suggesting emigration had occurred.

There is a broad agreement in the literature that selective hand harvesting of species with rapid recovery rates at small scales and following management best practice is unlikely to result in substantial adverse environmental impacts. However overharvesting has still occurred (for example of *H. elongata* in France and Portugal) and there are concerns that smaller red and green species may be entirely cleared from shore in Wales and England by intensive or cumulative harvesting activity. Due to slow growth, *A. nodosum* and *L. hyperborea* are considered the most vulnerable to unregulated or increased harvesting intensity.

5.3 Management approaches

Currently in Wales and England, commercial seaweed harvesting requires landowner permission, which for Crown Estate owned land involves a licence, only granted when a Competent Authority has assessed whether an adverse effect on site integrity can be ruled out on a SAC, SPA or Ramsar site. Recreational harvesting only requires landowner permission. Additional permission from statutory agencies is required for harvesting within designated sites. Similarly to commercial harvesting, this involves the assessement by a Competent Authority of whether there could be an adverse effect of harvesting on a SAC, SPA or Ramsar site and if it is within a SSSI, an assent may be required under Section 28H of the Wildlife and Countryside Act.

While the activity may be subject to regulation using legislation, this has not been implemented for hand harvesting of seaweeds in Wales and England to date.

Existing harvesting guidance documents are centred around Codes of Conduct which are not legally binding.

Identified management approaches are:

- Harvesting methods

  - Cutting height, leave a proportion of the plant (holdfast and some frond) remaining at the base;
  - Selectively cut with scissors rather than plucking or uprooting to support recovery and reduce by-catch;
  - Avoid by-catch of epiphytes and vulnerable species
  - Avoid harvesting reproductive material if possible (or only take half from each plant in the case of H. elongata);
  - For certain species (e.g. *for F. serratus* and *F. vesiculosus*), only harvest part of mature plants
- Harvesting period
  - Harvest during active growing season;
  - Avoid harvesting during the reproductive season;
- Harvesting frequency
  - Fallow periods for recovery of canopy in A. nodosum and perennial kelps
- Harvest limits
  - Quotas, volumes or bag limits;
  - Proportion of standing stock biomass removed/left remaining
- Harvesting spatial considerations
  - Harvest sparsely, leaving unharvested plants between those taken;
  - Shape, size and spacing of harvested areas (i.e. between harvested plants or patches);

Not specifically covered in guidance but considered useful are:

- Biosecurity planning for invasive non-native species and
- Community management and co-governance.
- No-take protected areas, including those designated by existing legislation (as a reference area, to protect a source population, or maintain ecosystem services such as biodiversity or coastal protection);
- Spatial, temporal and seasonal closures (e.g. in response to stock declines, during peak seaweed reproductive times, seasons of slow growth, or during breeding/nursery periods for associated marine life)

Effective management will need to be site and species specific. If followed or enforced, management can be effective in minimising the ecological impact of harvesting. Bennion *et al*. (2019) outline the need for baseline information and a rapid assessment technique to allow routine monitoring of wild resources in order to inform effective management. Monitoring of ecological baselines, such as the available standing stock biomass, and reporting of harvested quantities will be required in order to quantify the spatial and temporal extent of harvesting activities. At present, there are no estimates of standing stock biomass, or routine ecological monitoring on the impacts of small scale hand harvesting for Wales and England. Very small scale operations may not require monitoring (which is unlikely to be feasible due to the cost), while cumulative effects of multiple activities, or larger scale operations are likely to require some management to mitigate environmental impacts. Currently, management is hindered by a lack of sufficient information for most species to determine harvesting thresholds or site carrying capacity for harvesting activity.

Effective management strategies utilise a combination of management approaches. For example, Irish management of *A. nodosum* is considered a sustainability success. This involves cutting at a certain height (usually 20-25cm) above the holdfasts, rotating harvested areas with fallow periods of 3-5 years, and only taking a proportion (following a precautionary approach or informed by monitoring data) of the available standing stock.

As Codes of Conduct are voluntary measures, they require effective promotion and engagement to be a successful management tool. Whereas licences, quotas or rotational systems require enforcement and regulation, either nationally or through industry participation. Where areas are subject to intensive harvesting and cumulative pressures it is possible that more enforcement and regulation is required. For small-scale, low intensity harvesting, particularly where this is carried out by single or few companies with long-term interest in sustainability then Codes of Conduct and monitoring may be enough to protect the resource.

In the absence of legislative management, self-regulation can be effective when one business or harvester has exclusive access to a resource, which can potentially generate strong motivation for sustainability (e.g. Cornish Seaweed Co, Uist Asco and the Hebridean Seaweed Company). However there is currently no capacity to monitor the cumulative effects of multiple harvesters, which is a source of concern.

5.4 Monitoring approaches

Best sustainable practice advises estimations of crop standing stock prior to any commercial harvesting, in addition to assessments of density or percentage cover, reproductive and growth season of target species (Bailey & Owen 2014).

Reporting of harvested volumes by species is a condition of the Crown Estate licence, but is not required by most private landowners or for recreational harvesters. For effective marine planning, greater understanding of the long term effects of hand harvesting on standing stock and sustainability are necessary, which could be underpinned by records of harvested volumes. A record of harvested volumes, locations and effort per species could therefore be maintained by NRW and NE to address this gap.

To achieve sustainability for selected species (i.e. those which are most vulnerable due to life history traits, or which are heavily targeted), a robust and extensive survey of standing stock and areal extent of targeted species should be conducted to establish a reliable baseline against which to monitor change. Following the initial survey, population-level data should be used to determine what level of sampling would be required in order to detect different thresholds of change (i.e. 10, 20, 50% loss). A variety of survey methods are available for monitoring, including shore-based surveys (along transects, grids or haphazardly as appropriate) on species' abundance, biomass and distribution, and remotely-sensed imagery (i.e. from satellites, aerial surveys or drones) appropriate for larger species. Species distribution models can be used to estimate the likely abundance and biomass of targeted species, and to extrapolate from survey sites to the wider region, indicating areas likely to be vulnerable to harvesting impacts or other local pressures.

5.5 Knowledge gaps and how these can be addressed

Evidence was not found on the impacts of hand harvesting for all species, and examples from Wales and England were relatively limited. There was a good body of evidence found relating to recovery rates of the canopy biomass of large, charismatic seaweed species such as *A. nodosum* and *L. hyperborea*. With the exception of *C. crispus*, which has been well studied in the western Atlantic (from which conclusions should be applied to the UK with caution due to observed differences in growth rate), there was more limited evidence on the recovery rates of many smaller species. Some of these such as *M. stellatus* which are longer lived could be of concern. Population level impacts have been documented for *A. nodosum, L. hyperborea* and *C. crispus* in response to mechanical harvesting, which could be considered analogous to intensive hand harvesting in certain cases, but no evidence was found for these effects resulting from hand harvest. A recent PhD thesis has addressed knowledge gaps for *P. dioica* in South Wales, towards a goal of sustainable cultivation (Knoop, 2019).

Baseline ecological data, wild standing stock biomass, and potential stock volumes available for sustainable harvested are absent for most species (with the exception of *A. nodosum* for which the values represent Scottish stocks – Burrows 2010). For species such as *Porphyra* spp. with a long and locally intensive history of harvest, an unharvested “natural state” may be impossible to discern (Perry *et al*., 2014).

There is very little published information of harvested volumes from Wales and England, both commercially and for recreational use, with the latter being very difficult to quantify, and the former frequently subject to commercial confidentiality.

The majority of monitoring studies are spatially or temporally limited, focussing on either a small local area, from which it is difficult to extrapolate regionally, or of restricted duration to observe medium-long term impacts. For example, *F. vesiculosus* was still present within the previously monospecific *A. nodosum* canopy 12 years after harvest (Sundene 1973; Choi & Norton 2005), but it is not known whether this was a permanent phase shift or whether *A. nodosum* eventually returned to dominance.

These knowledge gaps reflect that applied science is a lower priority for research funders. A typical PhD project is three years and funding timescale tend to reflect this with funding periods typically for three to four years. Funding long-time series studies is challenging and hence most studies are short-term. Seaweed harvesting in the last few decades has been small-scale, increased interest and intensity is likely to stimulate research and management interest and increase the priority of studies that can answer key management issues.

Where possible this review has drawn on information from other countries but these should be interpreted with caution as they may not be directly applicable to UK shores.

Management knowledge gaps

There is good evidence for application of cutting height and fallow periods to *A. nodosum* harvest from Ireland and Scotland, however less literature was available on use of these approaches for other species.

Limited evidence was found for measures such as harvest limits (quotas, volumes or bag limits) shape, size and spacing of harvested areas, spatial, temporal and seasonal closures, by-catch volumes, no-take protected areas, community management and co-governance specific to hand harvesting from the UK.

Very limited data were available for standing stock biomass of most species. Further, there are no checks in place to ensure that reporting of harvested volumes (required by the Crown Estate licence) is fulfilled and in practice statutory nature conservation bodies do not always receive full detail of harvested volumes. Although recovery rates of some seaweed populations and their associated communities have been published, findings show considerable variation, and limited examples were available from Wales and England. Further, there was little information of variations in recovery rates under different environmental conditions, or differing harvesting scale / intensity. For many species, dispersal potential is poorly understood, and life-cycle processes which underpin recovery (for example in *Porphyra* spp.) have only recently being elucidated. Monitoring studies which investigate the effect (both on the seaweed stock and ecological functioning) of hand harvesting regime are needed for most species, with the exception of *A. nodosum*.

Biomass estimates are a key knowledge gap for almost all species harvested in Wales and England . With the exception of *A. nodosum*, for which management practices in Ireland are well established, and to some extent *C. crispus* (although caution should be exercised in generalising to Wales and England from *C. crispus* management examples in the western Atlantic),

There is a clear evidence gap as to what would be considered an appropriate or significant volume to harvest for seaweed species in Wales and England. This is particularly the case for small red and green seaweeds. Without this evidence it is difficult to advise threshold levels above which regulation or management should be implemented.

There is abundant information, mainly from Norway, on sustainable kelp harvesting practices which have been developed over decades. However as these utilise mechanical processes they are of limited application to hand harvesting. The impact of hand harvesting on British kelps has not been studied.

Interactions between seaweed harvesting and other processes such as the spread of non-native species and climate-driven changes in seaweed distribution and performance remain to be clearly understood. Specifically, if kelp forests become more fragmented and disturbed, could it lead to an increased abundance of *S. polyschides* (as has been observed in France) or facilitate the spread of *U. pinnatifida* and/or *L. ochroleuca* further into Wales and England .

Relatively little information exists on the extent to which harvesting canopy-forming seaweeds impacts wider fisheries production. Therefore, the links between fisheries habitat (provided by seaweeds) in Wales and England with other stakeholders (i.e. crab fishers) remains a significant knowledge gap.

Finally, while ecosystem service provision by certain seaweed species (particularly kelps) is well studied, there is very little evidence for most of the smaller intertidal species. The effect of seaweed harvesting on hydrodynamics, wave attenuation and consequences for coastal protection and erosion warrants further investigation for larger and more structurally complex species. The role of seaweeds (particularly highly productive kelps) in carbon storage has been largely overlooked until relatively recently. It is entirely unknown how seaweed harvesting would affect carbon cycling in coastal waters, but intuitively it is likely to reduce total carbon fixed and donated, at least at local/regional scales.

5.6 Recommendations

Well-structured management plans can help mitigate the impacts of harvesting and ensure sustainability of the resource, especially when developed in collaboration with scientists, nature conservation authorities and harvesters.

Design of a sustainable harvesting strategy is complex and must be site and species specific. Strategies should take into account factors such as the temporal variability of the resource, current and future volumes required by harvesters and the cumulative effect of multiple harvesters which will intensify pressure. A new guidance document is currently under development for Pembrokeshire Coast National Park (Bunker, 2021). The approach will utilise a decision making tree to formulate bespoke guidance specific to seaweed species and geographic area. First the potential impact of harvesting on the features for which a site is designated will be assessed, for example intertidal trampling or erosion of sand dunes for site access. Harvesting which impacts on designated features is considered inappropriate. Secondly, areas of concern, such as effects on the associated community, will be identified on a site and species specific basis. The tool will be modifiable and adaptable to emerging knowledge on standing stock biomass, impacts, or local species declines. When finalised this may provide a useful example that could be adapted for other sites. The process developing such tools is likely to be relatively site-specific as it should consider factors such as other sensitivie habitats and species present.

Best sustainable practices include estimations of crop standing stock prior to any commercial harvesting, and a robust monitoring strategy to inform management. The specific approach to monitoring will be dependent on a number of factors, including proposed harvesting intensity, characteristics of the site and wider region, targeted species, management objectives and available resources. To address evidence gaps around biomass, models predicting seaweed biomass and distribution could be applied to Wales and England, these would be useful for contextual decision making regarding wider ecosystem effects. In addition, accurate measurements of seaweed standing stock should be conducted in specific proposed harvesting locations as sustainable biomass is more relevant on a case by case basis. Biomass limits should consider the species of seaweed being harvested, intensity, frequency, harvesting method and location. Other considerations are also likely to apply on a site-specific basis. For example harvesting may be limited to reduce disturbance to birds or seals.

In general, for a relatively small biomass (no more than a few wet tonnes) that is not occurring over a very small area, a Code of Conduct may be appropriate to deal with what are expected to be limited impacts. The final decision to request a stock biomass assessment for greater amounts could be applied flexibly depending on species, location, etc.

Recording and reporting should be standardised, and commercially harvested volumes submitted via a national governing body to the FAO. Development of an accessible, standardised monitoring protocol (e.g. submission of photos) to investigate harvesting impacts is recommended (McLaughlin 2006).

To avoid overharvest for recreational use at popular sites, management strategies must include capacity to monitor cumulative effects. A public register of activities could help to identify sites of intense harvesting activity which may be at risk of cumulative effects

(Scottish Government 2016). While it is acknowledged there may be issues around administration, it may be possible for such an initiative to be incorporated into self-regulation processes by industry. Commercial harvesters should be required to keep records of harvested volumes, areas, timings and effort, in order to monitor for declining stocks.

Contingency for unproductive years resulting from poor weather or other factors, must also be incorporated, without encroaching into areas which are set aside as source populations for seaweed propagules or protected areas for associated biodiversity.

No-take areas appears to be effective as part of a comprehensive management strategy, and should be considered for broader management, particularly within protected sites in order to maintain favourable status.

Recommendations for management of slow growing *A. nodosum* are a 3-5 year fallow period between harvests, an increase in cutting height restrictions in line with Ireland and Scotland (20-30cm), and that the total allowable harvest per year should be only a fraction of the total standing stock available within economically-viable distance from a landing point, where biomass density is sufficient to justify a harvest (Burrows *et al.,* 2010).

Certain sites may require additional, specific management attention, such as Freshwater West and Bracelet Bay, due to the intense collection of Laver which has been identified as a concern (Perry *et al.,* 2014). Although no similar site specific evidence was found for areas considered to be of high risk in England.

The distribution maps and information on range in the Appendices will support managers to understand which species may be at risk from climate change. For example, *A. esculenta* populations are declining in England, therefore harvesting of remaining stocks may need additional management considerations.

Biosecurity plans should be developed alongside management plans to ensure measures are applied to limit the spread of INNS. These should include recommendations to avoid harvesting individuals that are heavily fouled, prevent return of removed INNS to the environment and limit spread of INNS. Harvesters should ensure that equipment is checked, cleaned and dried between sites.

6. Conclusions

The impact of seaweed harvesting activities will reflect management strategy (influencing the frequency, intensity and proportion of biomass removed), and extent of compliance / enforcement as well as the result of regional, seasonal, and species specific variations in growth rate and recovery capacity of both the seaweed and their associated community. As such, effective management is likely to be site and species specific. Due to gaps in evidence, it is anticipated that some difficulties will still remain to staff giving advice on harvesting. This report has identified management approaches that are well-supported and based on ecological considerations such as life-history and recovery mechanisms. The detailed species appendices provide the evidence base to support managers in developing species specific management plans.

For effective management, greater understanding of the long term effects of hand harvesting on standing stock and sustainability are necessary. Unregulated hand harvesting has been documented in the literature and concerns have been raised for the conservation of some UK species due to the increasing interest in harvesting. Although small scale, the future of unregulated seaweed harvesting activity in Wales, particularly for Laver and fertilizer, has been identified as a concern (Perry *et al.,* 2014), which requires monitoring and reporting. Cumulative effects of multiple harvesters require management attention, and could be addressed through the creation of a harvesting register.

Effective management will be site and species specific, utilising a combination of management approaches. Monitoring of ecological baselines and reporting of harvested quantities will be required in order to quantify the impact of harvesting activities. Management plans should also be based on site-specific conditions and take into account the wider ecosystem and the presence of particularly sensitive habitats and species. Key knowledge gaps remain regarding the lifecycle and recovery capacity of certain species, particularly red seaweeds and the standing stock biomass of all species available for harvest. However, seaweed harvesting activities provide an opportunity to involve stakeholders to conduct applied research projects, monitoring sites before, during and after harvesting and to trial effective management issues. Continued monitoring in some form by harvesters could be a condition of any license to exploit wild stocks. Stock assessments by harvesters would probably work best where areas are licensed to a single harvester or company and thereby a long-term interest in maintaining the stock.

Glossary

Apical meristem The growing point at the tip of the axis.

Basal Forming or belonging to a bottom layer or base.

Blade Flattened part of a seaweed that resembles a leaf.

Bycatch A fish or other marine species that is caught unintentionally while fishing for specific species or sizes of wildlife. Bycatch is either the wrong species, the wrong sex, or is undersized or juveniles of the target species.

Calcified seaweeds a calcium-carbonate encrusted red alga which produces calcareous prolongations. These include maerl (*Lithothamnion glaciale* and *Phymatolithon calcareum*) which accumulate subtidally forming dense beds of calcareous material, both living and dead. Intertidally harvested calcified seaweed is likely to comprise *Corallina officinalis*.

Conchocelis In the life cycle of Porphyra, a haploid stage preceding the stage that is recognized as the gametophyte.

Ephemeral Lasting for a very short time.

Epiphyte Organisms growing on the surface of a living plant (but not parasitic upon it).

Ecotypic A genetically distinct population of plants, animals, or other organisms that is found in a particular habitat

Frequency how often the seaweed is cut in terms of harvesting intervals, period between harvesting.

Frond Term used to refer to stipe and blade together.

Gametophyte A gametophyte, or gamete-bearing plant, is one of the two multicellular phases that occur in alternation of generations. The gametophyte is the haploid phase

Green seaweeds smaller, simpler in structure and shorter-lived than wracks and kelps green seaweeds generally, range from a few centimetres to a metre in length. Key species include *Ulva intestinalis* and *U. lactuca*.

Hand cutting or picking This method involves harvesting living species by hand at low tide using tools such as serrated sickles or scythes.

Hand gathering This method involves the collection of beach-cast species from the strandline by hand.

Haptera Plural of 'hapteron'.

Hapteron: Root like structure of macroalgae holdfasts.

Holdfast: Base of a seaweed that attaches it to a rock; this resembles roots

Intensity: the amount of seaweed harvested from a given area with a given frequency. i.e. high intensity would remove the entire standing stock biomass from an area of shore and be repeated with regular frequency.

Kelps: large brown seaweeds of the taxonomic order Laminariales. British kelp forests mainly comprise *L. hyperborea*, but also include *L. digitata, A. esculenta, S. latissima* and *S. polyschides.*

Mechanical 'hedge' cutting: Specialised vessels called mechanical seaweed harvesters that work close to the shore and cut the living seaweed as the stalks float above the seabed. These vessels include the Norwegian suction/cutter harvester which is designed to harvest *A. nodosum*.

Mechanical gathering: This method involves the collection of beach-cast species from the strandline using tractors or JCBs. 3

Meristem A type of tissue found in plants. It consists of undifferentiated cells (meristematic cells) capable of cell division. Cells in the meristem can develop into all the other tissues and organs that occur in plants.

Photoperiod The period of time each day during which an organism receives illumination; day length

Primary production: The process of synthesising organic compounds from inorganic carbon (principally carbon dioxide). The fixation of carbon principally occurs through photosynthesis, which uses light as its source of energy. Primary producers (in marine environment mainly algae) form the base of the food chain.

Proportion: the proportion of the individual seaweed plant harvested (i.e. 50% of the blade), or the proportion of the overall population removed in a harvesting event.

Red seaweeds: smaller than brown seaweeds and kelps with a similar size range to the green seaweeds. This group includes perennial species: *C. crispus, M. stellatus* and *P. palmata* and annual *Porphyra* spp.

Sorus A cluster of spores

Spore Typically one celled reproductive unit capable of giving rise to a new individual without fusion with another reproductive cell.

Sporophyll Additional structures, produced by some kelps, above the holdfasts and below the blade, which resemble small thicker blades or may be flattened outgrowths from the stripe.

Sporophyte The diploid, spore producing, asexual generation in the life cycle of some plants; typically formed by the fusion of haploid gametes.

Stipe Stalk of a seaweed between holdfast and blade

Substrata Material available for colonization by plants and animals

Trawling/Sledging/Dredging: In the case of kelps, this involves a device which tears living plants larger than a certain size from the substrate and leaves smaller plants for re-growth (i.e. generally only mature plants are harvested). Existing devices include the Norwegian kelp dredge designed to harvest *L. hyperborea* and the Scoubidou which is designed to harvest L. digitata. These devices operate in areas of rocky substrate and therefore differ from other forms of dredging (e.g. scallop dredging) that physically disturb the underlying substrate. There may, however, be some potential for physical disturbance of the substrate by other devices (e.g. dredgers used in maerl extraction).

Volume: the amount (in tons) of seaweed harvested over a given time frame.

Wracks (rockweeds): large brown seaweeds of the taxonomic order Fucales. Species include *A. nodosum, Pelvetia canaliculata, F. vesiculosus, F. spiralis, F. serratus* and *H. elongata*.

Zoospore: A spore which is capable of locomotion

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Appendix 1 *Alaria esculenta*

Names

Alaria esculenta

Atlantic Wakame, Dabberlocks, Wing(ed) kelp, Honeyware, Scotland: Keys; Ireland: Murlins, Lair, Laracha, Ribini, Cupog nag cloc

Alaria esculenta: harvesting

A. esculenta is hand harvested, usually with a knife at low tide. Harvesting in Wales and England is likely to occur similarly to Scotland during a narrow time window in the early summer. Juvenile plants are avoided by hand harvest (Scottish Government *et al.,* 2016).

The species has a long history of use in the UK, and its name *A. esculenta* literally means 'edible wings'. Traditionally, *A. esculenta* was used in both Scotland and Ireland for human consumption, animal feed, and fertilizer (Newton, 1931; Guiry & Hession, 1996; Guiry, 1997; Guiry & Blunden, 1991). More recently, the species is also used for alginate production, bodycare products (Guiry & Blunden, 1991; Guiry, 1997), and aquaculture feed production (Mai *et al*., 1996).

Young *A. esculenta* can be eaten as a substitute for *U. pinnatifida* (Wakame), hence one of its common names “Atlantic Wakame”. In North America especially, *A. esculenta* and *A. marginata* are rapidly gaining popularity in the natural foods market (Lewallen & Lewallen, 1996). It is rich in sugars, proteins, vitamins and other trace metals and contains up to 42% alginic acid (Levring *et al*., 1969; Indergaard & Minsaas, 1991; Lewallen & Lewallen, 1996).

A. esculenta is also cultivated at a pilot scale in parts of the UK, particularly Ireland and Scotland.

Alaria esculenta: Natural range / distribution

A. esculenta is usually found from the low water into the subtidal to about 8 m depth on exposed to extremely exposed rocky shores, where it grows on hard substratum (bedrock, boulders, and cobbles). It is tolerant of a temperature range from -2°C to 16°C, tidal flows between weak < 1 knot (<0.5 m/sec.) to strong (3 to 6 knots, 1.5-3 m/sec.), and requires full salinity.

Kelp morphology is highly variable according to environmental conditions. *A. esculenta* exhibits a short stipe and narrow frond base in exposed conditions whereas in sheltered conditions the stipe is long and the frond base wider (Widdowson, 1971). It is not known if these variations affect desirability for harvest.

The British distribution is from the Shetland Isles, Orkney and east coast of Scotland, south to Flamborough Head in England, continuing along the South west of England and the west coasts of England, Wales, Scotland and Ireland including the Isle of Man. It is largely absent from most of the east coast of England, due to a lack of suitable substrata (See Figure 1).

It is absent in the southern North Sea and English Channel is due to high summer surface temperatures of 16 °C, which it cannot survive (Munda & Luning, 1977; Widdowson, 1971; Sundene, 1962). Declines of this species have been anecdotally reported in England, Wales and Ireland, which are thought to be driven by climatic warming.

A. esculenta occurs in the North-east Atlantic from Novaya Zemlya in the Barents Sea to Iceland, and south to Brittany. Globally it also occurs from the shores of Greenland to the Bering Strait, in the Bering Sea and Sea of Japan in the North Pacific.

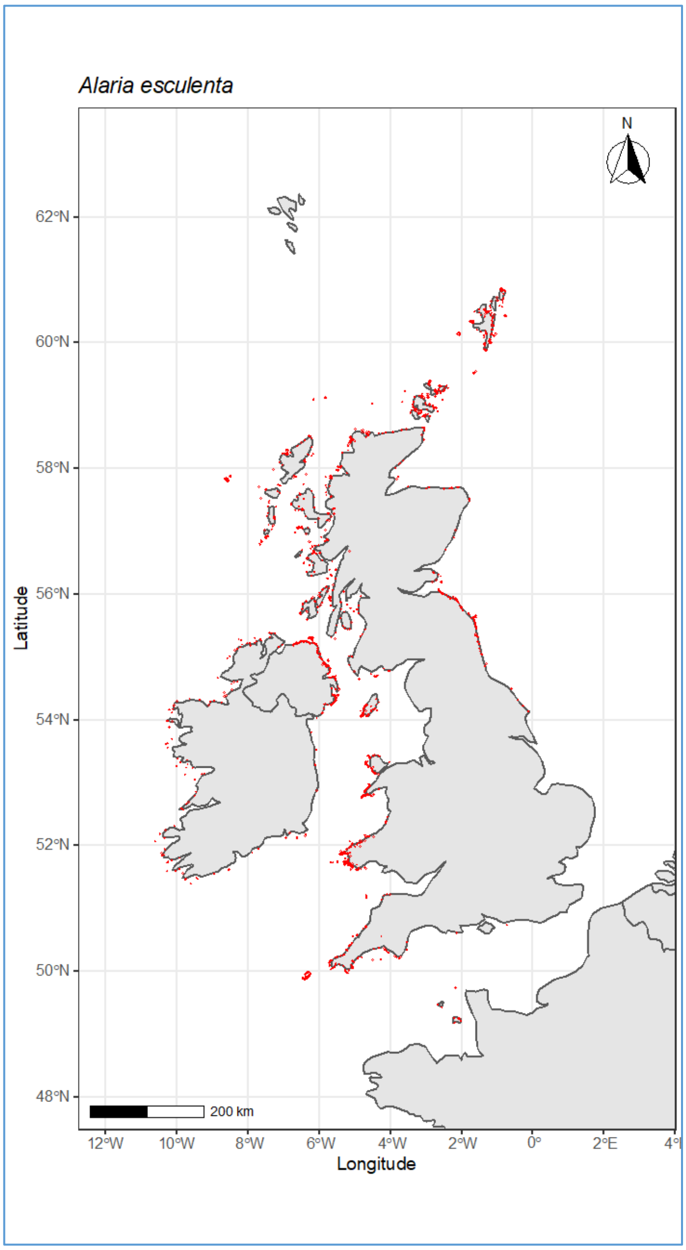

Figure 1. Distribution of *A. esculenta* in the United Kingdom and Ireland. Occurrences of *A. esculenta* plotted as red points. Spatial data of species occurrences obtained from NBN Atlas, DASSH and Marine Recorder databases.

Alaria esculenta: Growth, reproduction and seasonality

A. esculenta may reach 4 m in length, in conditions of strong currents and low wave action (e.g. Aran Islands, Ireland; Guiry, 1997; Werner & Kraan, 2004), with fronds reaching a total length of 2m, however, growth rates are locally variable and are more typically 30-90 cm in length (Birkett *et al*., 1998b). It is a perennial maturing at between 8-14 months old, and may live for 4-7 years (Birkett *et al*., 1998b; Baardseth, 1956).

Growth is seasonal, with spores released between November and March and juvenile plants (sporophytes) appearing in early spring (Birkett *et al*., 1998b). Maximum growth rates are recorded in April-May which can exceed 20 cm/month (Birkett *et al*., 1998b) (although an average of 5 cm and 10 cm per day have also been reported (Birkett *et al*., 1999b; Kain & Dawes, 1987). From June-July growth rates slow and continual erosion, especially in winter months, can reduce the blade to just the midrib. As such, harvesting activity is likely to occur over a narrow window in late spring to early summer, to obtain maximum yield before senescence begins (Scottish Government *et al.,* 2016).

The lifecycle alternates between a visible sporophyte (the seaweed 'plant' see on the shore) and microscopic, filamentous gametophyte stages (Van den Hoek *et al*., 1995). Reproductive "sporophylls" form on the stipe during spring and in a lesser amount in autumn only (Kraan pers. comm, Widdowson, 1971). When mature, specialised sorus tissue forms on the sporophylls, from which large numbers of spores are released between October and May (Kraan pers. comm.; Birkett *et al*., 1998b). The spores remain in the water column for a period of 24 hours before settling onto the seabed (Birkett *et al*., 1998b). These spores develop into gametophytes, which become fertile in under 10 days in optimal conditions. The gametophytes produce gametes (sperm and eggs) which fuse after fertilization, forming juvenile platelets (germlings) in situ of the female gametophyte. Juvenile sporophytes appear in early spring.

If conditions are not optimal, the gametophytes can develop vegetatively, forming an invisible "seed bank" in the understory, which may persist for several years. Fragments of damaged vegetative gametophytes may develop into separate gametophytes (only a few cells are required) hence reproductive potential may be increased. If optimal conditions return the gametophyte may become fertile and produce gametes. However, successful fertilization requires a high density of spore settlement (about 1 mm apart).

Alaria esculenta: Recruitment and recovery potential

A. esculenta is an opportunistic colonizing species (Kain and Jones 1975; Hawkins & Harkin 1985; Hill 1993) which will settle on bare surfaces including mobile boulders. It often appears early in the algal succession (ca 3 months after clearance of dominant algae) although it is out-competed by other kelp species on less wave exposed shores.

During kelp canopy removal experiments in the Isle of Man, Hawkins & Harkin (1985) found that in moderately wave exposed areas cleared of *L. digitata* (the dominant canopy forming species) *A. esculenta* became the dominant canopy algae within 9 months (October - June) and *L. digitata* did not re-establish dominance within the study period of 15 months. In areas of moderate to sheltered wave exposure *A. esculenta* colonized the blocks within 1 month of clearance and reached 25% coverage within 5 months but within 7 months *L. digitata* had out-competed *A. esculenta* and re-established dominance within the community reaching ~90-95% coverage.

A. esculenta may have a lower dispersal capacity than other Laminarins due to the location of the sporophylls at the base of the plant close to the substratum (in contrast with Laminarian kelps, which release spores from the blade higher up into the water column), (Norton 1992; Sundene 1962). The dispersal potential of zoospores may be <2m, although settling rate is dependent on local currents, therefore spore settling time is probably longer than 1 day. Gametophytes may disperse between 1 -10 m by drifting, maybe up to 100 m, and if an entire mature adult plant (sporophyte) is dislodged complete with it's reproductive sporophylls, there is the potential that it may drift a substantial distance releasing spores.

Recruitment is likely to be influenced by the proximity of mature plants producing viable zoospores (Kain, 1979; Fredriksen *et al*., 1995).

There is considerable material on the effects of harvesting kelp species (Birkett *et al*., 1998; Guiry & Blunden, 1991) but little evidence concerning the effects of harvesting on *A. esculenta* populations. In summary, evidence from other kelp species suggest that the canopy can recover within 3-4 years, although the effects on the rest of the community is poorly studied. In clearance experiments, *A. esculenta* may appear early in the succession suggesting that it would recover relatively rapidly.

Alaria esculenta: Ecological importance

Relative to Laminarial kelps, *A. esculenta* is fast growing and fairly short lived, which will limit the opportunity for a diversity climax community to develop in association with this kelp, however it's ecological importance is still considerable.

Primary production and nutrient cycling

Due to its rapid growth and high productivity this species is likely to play an important role in the turnover of organic matter, export of detritus, carbon capture, and nutrient cycling however no literature specific to *A. esculenta* was found within the time frame of the review.

Habitat provision-epiphytes

Bryozoa and several epiphytes including *Litosiphon laminariae* (Kraan pers. comm.) colonise the fronds, and spore production may be inhibited by epifauna such as *Membranipora membranacea* (sea mat) and endophytes such as *Streblonema* sp.

Habitat provision- holdfasts

In the extremely exposed Rockall *A. esculenta* forests, the holdfasts become covered in a thick layer of encrusting coralline algae and, after the death of the alga, the holdfast rots leaving a space under the coralline crust that is a habitat for several mobile species that would most likely not survive on the open rock in such a wave exposed situation (K. Hiscock, pers. Comm.).

Chitons were reported to be an important controlling agency in *Alaria* spp. Populations allowing an otherwise weaker competitor to dominate in the north east Pacific (Paine 1980).

Habitat modification (including shading and moisture retention)

A. esculenta forms the main canopy in exposed rocky areas, and is the dominant macroalgae in several biotopes (Lewis, 1964; Connor *et al*., 2004). It is thought that the sweeping action of *A. esculenta* fronds in wave exposed conditions prevents colonization by ephemeral algae by abrasion (Hawkins & Hartnoll 1985). However, *Mytilus edulis* settlement has been found to be significantly higher in close proximity to *A. esculenta* and is thought to increase beneath *A. esculenta* canopies (Bégin *et al*., 2004). Therefore, any loss of *A. esculenta* as a result of commercial extraction may dramatically affect the understorey community.

Alaria esculenta: Ecological implications of harvesting

No studies examining the effect of commercial extraction of *A. esculenta* were found. Stamp and Tyler-Walters (2015) and Tyler-Walters (2008) developed generic sensitivity assessment of *A. esculenta* to pressures associated with human activities.

Habitat modification (including shading and moisture retention)

Removal of the algal canopy would expose the understorey fauna and flora to increased desiccation. Experimental canopy removal experiments conducted in the Isle of Man (Hawkins and Harkin, 1985) found that following the removal of the canopy the understorey encrusting red algae became bleached and died within a week.

Relevant studies or monitoring

No relevant studies were found.

Species specific management approaches

- Harvest the blade 20 cm above the stipe (Edwards *et al*., 2012)
- Avoid harvesting the fertile material, located on the sporophylls at the base of the stipe
- Avoid harvesting during the reproductive season in autumn and winter

Access to this seaweed resource from the shore is limited by the tidal window, which could act as a buffer against excessive harvesting. However, should use of snorkel or SCUBA increase, harvesting pressure would increase correspondingly, potentially requiring additional management.

Alaria esculenta: Knowledge gaps

- Standing stock biomass
- Sustainable harvest volumes
- Ecological impacts of harvest

- Recovery time

Alaria esculenta: References

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Appendix 2 *Ascophyllum nodosum*

Names:

Ascophyllum nodosum

Asco, Knotted wrack, Egg wrack, Knotted kelp, Rockweed, Norwegian kelp, Yellow Tang, Sea Whistle, Irish: Feamainn bhuí

Ascophyllum nodosum: harvesting

Due to effective accumulation of nutrients from seawater, *A. nodosum* is primarily harvested as a food source for humans and seaweed meal for livestock. A high content of vitamins and minerals means it can be used as a dietary suppliant to improve dietary deficiencies (Pereira *et al.,* 2020).

A. nodosum is generally hand harvested by means of scissors, knives or sickles. The floating canopy is also hand raked at high tide from shallow draft boats in Ireland. It is also mechanically harvested in Scotland and the Western Atlantic.

Companies in the UK known to hand harvest *A. nodosum* are: Hebridean Seaweed Company, Böd Ayre, Mara, Irish Seaweed, Viking Fish Farms, and Seagreens harvest seaweeds (AB-SIG 2013). The species is recreationally and commercially harvested in Wales and England, but not to the same extent as in Ireland, Scotland, France, Norway, Canada and the United States.

In Ireland 25,000 wet tonnes annually are hand harvested (Edwards *et al.*, 2012). Wild harvest in the UK in 2013 has been estimated at around 2,000-3,000 dry tonnes (AB-SIG 2013).

Ascophyllum nodosum: Natural range / distribution

A. nodosum occupies the mid to lower shore and generally favours sheltered shores, where it grows attached to bedrock, boulders or cobbles (Hill & White, 2008). On more exposed shores, *A. nodosum* can be outcompeted by *Fucus* sp.

It is only present in the Northern Atlantic where it exhibits significant life history plasticity and the ability to withstand very large environmental variation (Svensson *et al.*, 2009) (See Figure 2). It's southernmost population currently occurs on the northern coast of Portugal (Borges *et al.,* 2020) whilst Greenland, North Norway and Northern Canada host its most northern populations (Marbà *et al.*, 2017).

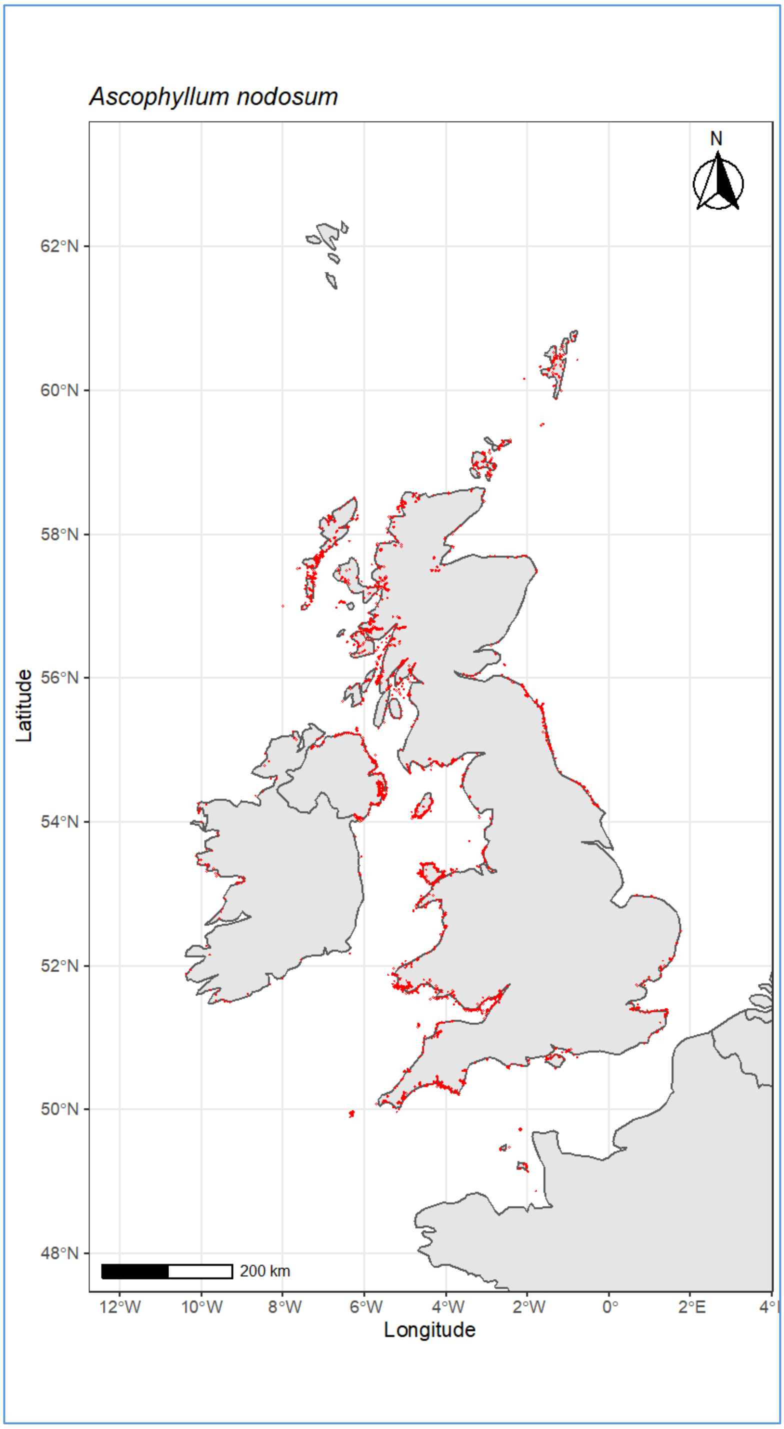

Figure 2 Distribution of *A. nodosum* in the United Kingdom and Ireland. Occurrences of *A. nodosum* plotted as red points. Spatial data of species occurrences obtained from NBN Atlas, DASSH and Marine Recorder databases.

Asophyllum nodosum: Growth, reproduction and seasonality

A. nodosum is a long lived, perennial species. The holdfast has been demographically modelled to last centuries (Åberg, 1992), with individual fronds lasting for approximately 10-20 years (Stengel & Dring, 1997), and clumps of plants reported being 60 - 550 years old (Seip 1980; Åberg 1992). The average age within populations of *A. nodosum* is high, and there is little population turn over (Schiel & Foster, 2006).

Growth occurs from the tip of the blade, with the yearly formation of a bladder or vesical (Marbà *et al.*, 2019). It is therefore possible to record annual growth increments by counting the bladders. Initially, growth is slow, approximately 0.2 cm in the first year. This rate increases to 1.5cm in the second year, and further with maturity (Sundene, 1973). A year's growth length is typically 8-15cm for mature individuals (Kelly, 2001).

Growth is seasonal, being lowest in November and December reaching maximum rates in the spring/summer (Stengel & Dring, 1997). Daily growth rate has been documented to be maximal in the morning, declining throughout the day (Strömgren & Nielsen, 1986). This suggests it may be better practise to only harvest mature individuals, and to avoid harvesting during the spring/summer months, as these are times of maximal growth.

Reproduction

A. nodosum takes five years to become sexually mature (Sundene 1973). *A. nodosum* can reproduce sexually or by vegetative propagation, where a new individual develops directly from the basal shoots of the parent plant. vegetative propagation is the most common and important method of reproduction for this species (Aberg 1996).

Mature individuals reproduce annually by gamete release and external fertilisation. This typically occurs from March-April in the UK, at times as early as January or February (Hill & White, 2008). Males and Females occur as separate plants. Dispersal capacity is low, suggesting that re-colonization after a mass mortality events can be extremely slow (Perry & Hill, 2020). As many as 2.5 x109 eggs m2/year may be produced in a mature stand of *A. nodosum* (Åberg & Pavia, 1997). It is estimated that mortality exceeds 99.9% in the first year (Dudgeon & Petraitis, 2005), and that ca one hundred millionths of eggs survive to become 1.5-year-old recruits (Åberg & Pavia, 1997).

Ascophyllum nodosum: Recruitment and recovery potential

It is widely observed that populations primarily consist of large, mature individuals with few recruits (Vadas *et al.,* 1990; Dudgeon & Petraitis, 2005). Dudgeon & Petraitis (2005) found *A. nodosum* mortality during its first year exceeds 99.9%.

Recovery from experimental clearance

Numerous studies have concluded that *A. nodosum* take long periods of time to recover from removal, with cleared areas being dominated by *Fucus* sp (Bertness *et al.,* 2002; Jenkins *et al.,* 1999; Dudgeon & Petraitis, 2005; Cervin *et al.,* 2005; Ingólfsson & Hawkins, 2008). It has been found that even after 12 years of clearing, a mixed canopy of *Fucus* spp. and *A. nodosum* is present. This has been attributed to slow growth rates and reduced competitiveness of *A. nodosum* in mixed populations, leading to other Fucus sp. being more dominant (Sundene, 1973; Choi & Norton, 2005). Additionally, large clearings (~8m diameter) were found to be quickly colonised by competing species, whereas small clearings allowed *A. nodosum* to recover more effectively (Dudgeon & Petraitis, 2005), presumably by processes of vegetative growth from base of plants remaining nearby.

Germling survivorship has been attributed to the size of the cleared area (Dudgeon & Petraitis, 2005), inter and intraspecific density-dependent competition of germlings (Choi & Norton, 2005), predation by grazers (Baardseth, 1970; Lazo *et al.,* 1994), their susceptibility to sedimentation (Airoldi, 2003), and inability to tolerate desiccation at low tide (Brawley & Johnson, 1991). All of which are arguably promoted by the loss of algal canopy, which inevitably affects recruitment and subsequent growth of a population. Canopy presence can reduce mortality by reducing exposure to light and high temperatures and inhibiting colonisation of competing species, i.e. *Fucus* sp, Corallines. (Connell *et al.,* 2014). Canopies also increase habitat complexity allowing for refuge against grazing pressure (Dudgeon & Petraitis, 2005). It is therefore important to maintain a population canopy to increase chances of recruitment, further supporting the need of an effective minimum cutting height and correct management of harvesting frequency, intensity and volume.

Dispersal

It is widely accepted that fucoids (including *A. nodosum*) have a low dispersal capacity, which limits the speed at which the species recovers from partial die-back (Perry & Hill, 2020). Under experimental conditions, attachment success was poor at low velocity wave action, with 99% of 1-minute old young zygotes being displaced (Vadas *et al.*, 1990). Therefore, calm conditions typical of wave sheltered habitats are required for successful recruitment (Perry & Hill, 2020). Recruitment is also found to be 1 to 2 orders of magnitude higher under the adult canopy which essentially dissipates and buffers wave energy, theoretically increasing local settlement (Lamote & Johnson, 2008). It is necessary to take into consideration the location of populations when giving guidance on the volume, frequency and intensity of harvesting as environmental factors such as wave action can have further detrimental effects on a population if the canopy is compromised.

Recovery from experimental harvesting

Keser *et al.* (1981) recorded the levels of re-growth exhibited by *A. nodosum* and *Fucus vesiculosus* after experimental harvesting in Maine. Harvesting was simulated by cutting fronds to three different lengths, that is, frond removed to the holdfast, 15 cm from the holdfast and 25 cm from the holdfast. Recovery was found to be more rapid in estuaries (Keser *et al.*, 1981). Of the fronds which that were cut back to the holdfast, only those within sheltered, estuarine and grazer free conditions showed any re-growth. More mature *A. nodosum* fronds cut back to 15 cm and 25 cm within a sheltered site showed some re-growth, however, there were high rates of mortality. The lack of re-growth was suggested to be caused by a lack of functional growing points found towards the bottom of the frond in older individuals. Almost all (95%) of young *A. nodosum* individuals cut back to 15 cm and 25 cm regrew. In almost all populations repeat harvests resulted in lower biomass yields (Keser *et al.*, 1981).

Printz (1959) also carried out harvesting experiments where fronds were cut back to 25 cm, 15 cm and 5 cm from the holdfast. Individuals that had been cut back to 25 cm had an ‘abundance of new shoots’ and had grown to 30-35 cm in length after a year. Individuals that had been trimmed back to 5 cm showed almost no change a year after the harvesting event. When the 5 cm individuals where re-visited three years after the harvesting event they were still almost unaltered. The reasons for the lack of re-growth were attributed to the lack of regenerative tissue found in the older flesh further down the thallus (Printz, 1959). Slow re-growth occurs after harvesting from the holdfast (Baardseth, 1970), with complete or partial removal heavily limiting recovery potential (Phillippi *et al.*, 2014). There is considerable evidence to suggest that recovery times are reduced considerably if fronds are cut higher up the thallus (Phillippi *et al.,* 2014 and references therein). Keser *et al.* (1981) also noted that repeated annual harvest reduced biomass, suggesting staggering annual harvest between sites to prevent large-scale destruction of the resource.

Overall, *A. nodosum* has high egg and juvenile mortality rates, slow growth, and can take over five years to reach reproductive maturity. Small scale perturbations (e.g. frond removal; Keser *et al.*, 1981; Pocklington *et al.*, 2018) and small scale clearances (e.g. Cervin *et al.*, 2005; Jenkins *et al.*, 1999; Dudgeon & Petraitis, 2005) have been shown to affect the community significantly. Minor disturbances that result in the cutting of the frond only may allow regrowth within two to three years depending on remaining length. However, even small-scale disturbances similar to the clearance studies (i.e. the removal of small patches, flush with the substratum and/or including the holdfast, within the bed), may require over 12 years for partial recovery of the *A. nodosum* canopy and its associated community (Jenkins *et al.*, 1999; Cervin *et al.*, 2005).

Ascophyllum nodosum: Ecological importance

Primary production and nutrient cycling

A. nodosum undergoes epidermal shedding, whereby 10% of vegetative frond biomass enters coastal waters each year (Halat *et al.,* 2015).

Habitat provision- understorey flora and fauna

A. nodosum communities support high levels of biodiversity (Jenkins *et al.*, 1999; Pocklington *et al.*, 2018) and are characterized by a diverse understorey of red algae, together with sessile and mobile invertebrates (Jenkins *et al.*, 1999; Pocklington *et al.*, 2018).

Habitat provision epiphytes

The fronds themselves provide space for attachment of encrusting or sessile epifauna and epiphytic algae (Jenkins *et al.*, 1999; Pocklington *et al.*, 2018).

Habitat modification (including shading and moisture retention)

Bertness *et al.* (1999) found that the presence of an *A. nodosum* canopy reduced maximum daily rock temperatures by 5-10°C. It was also reported that water loss via evaporation was an order of magnitude less than that in areas where the fucoid canopy had been removed (Bertness *et al.*, 1999).

Commercially harvested species

Commercially important fish are associated with *A. nodosum* (i.e. Saithe: *Pollachius virens*, Atlantic cod: *Gadus morhua* and Atlantic herring: *Chupea harengus*), along with crustaceans (Pereira *et al.,* 2020).

Ascophyllum nodosum: Ecological implications of harvesting

Primary production and nutrient cycling

Analysis of data for a single year from 10 sites around Nova Scotia suggested that the removal of biomass of A. nodosum from coastal environments by harvesting was associated with a reduction in the amount of detrital material entering the food web (Halat *et al.*, 2015). However, the actual amount and impact has been debated (Garbary *et al.*, 2017; Ugarte *et al.*, 2017).

Habitat provision

Studies on the effects of commercial **harvesting** on faunal communities associated with *A. nodosum* have found that removal can reduce abundances of epifauna found on the un-harvested biomass (Jarvis & Seed, 1996, Phillippi *et al.*, 2014). Changes in *A. nodosum* have also been found to affect the large, mobile fauna such as crabs or grazing gastropods (Bertness *et al.*, 1999; Fegley, 2001; Jenkins *et al.*, 1999; Phillipi *et al.*, 2014; Pocklington *et al.*, 2018).

Phillippi *et al.* (2014) replicated commercial harvesting techniques in Maine, USA where *A. nodosum* fronds were removed 40.6 cm from the holdfast and the lowest lateral branch must remain with the holdfast (DMR, 2009). The experiment looked specifically at the effect of canopy reduction on infaunal species living within the soft sediments within intertidal rocky shores where *A. nodosum* was present. The experiment found that invertebrate species found living on and within sediments were not negatively affected by the harvesting activity (Phillippi *et al.*, 2014).

Habitat modification (including shading and moisture retention)

Pocklington *et al.* (2018) examined community disturbance after removal of 100%, 50%, 245% and 0% of *A. nodosum* fronds (but not holdfasts). They concluded that a pulse disturbance (frond removal) of 50% loss of fronds increased the temperature under the canopy significantly and decreased the abundance of mobile invertebrates such as *Littorina obtusata*. Sessile taxa such as *Osmundea pinnatifida* and encrusting corallines could withstand a 75% loss of fronds but declined by half if 100% were removed.

Changes in community composition

Additionally, canopy removal can result in a shift in community composition. Increased levels of light penetration due to canopy removal can cause bleaching and decline in red algal turf along with an increase in grazing limpets. Subsequent increases in limpet recruitment inhibits recovery to the pre-disturbed state (Jenkins *et al.*, 1999).

Relevant impact studies or monitoring

Removal of non-target species (by-catch)

Although *A. nodosum* supports a large array of species, only *Littorina littorea* has received attention as bycatch due to its commercial interest (Sharp *et al.,* 2006).

Biomass stock assessment

A. nodosum stocks in the Outer Hebrides were investigated and mapped in 2010 (Burrows *et al.,* 2010). Prior to that, the last thorough review of Fucus stocks, including *A. nodosum*, was produced in 1947 (Walker, 1947). Further to this there has been no recent estimates of the wild seaweed standing stocks in the UK (Capuzzo & McKie, 2016).

Species specific management approaches

A. nodosum is commercially harvested in Scotland, Ireland, France, Norway, Canada and the United States (Table below). Management of hand harvesting is not uniform, with certain countries having little or no legally binding guidance (Table 2). Regional approaches to management within countries also differ (i.e. within Canada).

Generally, hand-harvesting management guidance includes:

- Use sharp knife or sickle
- Cut above holdfast
- Rotate Harvest every 3/5+ years
- Harvest areas sparsely, leaving un-harvested plants between harvested ones

Avoid harvesting during reproductive months may be advised. Considering this season may begin as early as January it might be necessary to monitor the stock on a monthly basis and cease harvest once reproduction commences.

Cutting of the stipe should be above the holdfast with different countries recommending different lengths:

- Wales and England 10-20 cm
- Scotland 30cm above the holdfast
- Ireland 20-25cm

Both Scotland and Ireland suggest a recovery time exceeding 3-5 years, with Wales/England suggesting a minimum of 2 years (Table 2).

Table 9 *Ascophyllum nodosum* hand harvest guidance between countries.

Harvest Guidance	**Scotland** (*Scottish Government, 2016*)	**Wales and UK** (*NRW, 2018; Bailey & Owen, 2014*	**Ireland** (*Bruton et al., 2009*)	**Côtes d'Armor, France** (Mesnildrey *et al.*, 2012).	**Norway** (*Meland & Rebours, 2012*)	**Canada**
Time of year	All year	Avoid spring reproductive period	All year	All year	.	Should avoid reproductive season (May/June) Harvest in first part of season through to summer
Cutting Height	12" (30cm) from base	10-20 cm from holdfast	Cut above the holdfast using a sickle 25cm from base (Guiry and Morrison, 2013)	Cut above the holdfast using a sickle	Mechanical harvesting leaves 10cm above holdfast	Nova Scotia Sea Plants Harvesting Act specifies 5" above holdfast. Environment Canada recommends 10" (25 cm) above holdfast (*Vandermeulen, 2013*). Canadian Fisheries Act - 12.7 cm from the rock surface holdfasts not to be removed. Québec - 15 cm above holdfast (Gendron *et al.*, 2018)
Frequency	3-4 years rotation	2+ year rotation	Recovery time 3-4 years. Fallow period of 4-5 years (*National*	2 year resting period. In other territories seaweed gatherers	Harvest seaweed beds every 4-6th year to ensure re-growth	Nova Scotia: ? Québec- 3 year recovery period (Gendron *et al.*, 2018)

Harvest Guidance	**Scotland** (*Scottish Government, 2016*)	**Wales and UK** (*NRW, 2018; Bailey & Owen, 2014*	**Ireland** (*Bruton et al., 2009*)	**Côtes d'Armor, France** (Mesnildrey *et al.,* 2012).	**Norway** (*Meland & Rebours, 2012*)	**Canada**
			Seaweed Forum, 2001)	impose rotation system for the fields themselves		
Intensity	Circle subsection of adult population with rope with unharvested areas in-between	Harvest small areas with unharvested areas in-between	~75,000 tonnes could be sustainably harvested each year, less than half the natural resource is being exploited (*Hession et al., 1998*)	Total landings are fixed at 3 000 tons per year in Côtes d'Armor.	No official regulations on harvesting of A. nodosum No impact assessment or documentation is required by the authorities Needs permission from landowner. Agreement is met between landowner and harvester.	Nova Scotia -Production estimated to be between 0.61 and 2.82 kg/m² (Cousens, 1984). Estimated an annual production in south west Nova Scotia of 2.0 - 2.6 kg/m² for a standing crop of 8kg/m² by assuming recovery from harvesting takes 3-4 years (Hill and White, 2007). Québec - 17% per year of the harvestable biomass (Gendron *et al.,* 2018)

Conclusion

It is necessary to keep the holdfast intact and cut the *A. nodosum* ~25 high so to allow re-growth. Re harvest is then possible after 3+ years. If the whole crop is removed recovery will be slow due to its poor recruitment and slow growth rate. Along with the recovery of algae stock, it is also necessary to take into account canopy maintenance in terms of ecological structure and function. Spare harvest within any given area will minimise the effect of habitat loss and other services such as coastal protection. It is necessary to monitor by taking regular pre and post harvest stock assessments from which inference can be made about the recovery of associated communities.

Ascophyllum nodosum: Knowledge gaps

- Although well studied, the majority of studies focus on impacts of harvesting on the seaweed resource rather than impacts on wider ecosystem
- English and Welsh baseline distribution and standing stocks
- Demography of early life stages
- Few experimental studies have examined survivorship
- Research is required to provide evidence for environmental sustainability
- Standardisation of guidance for the whole of UK

Ascophyllum nodosum: References

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Appendix 3 *Chondrus crispus*

Names

Chondrus crispus

Irish moss, Carrageen, Jelly moss, Irish: Carraigin, Fiadham

Chondrus crispus: harvesting

Together with *M. stellatus, C. crispus* is harvested commercially for extraction of the phycocolloid, carrageenan, for pharmaceutical, cosmetic and food industries (reviewed by Guiry & Blunden, 1991). Artisanal uses include boiling with milk and sugar or honey to serve as a drink, and as a remedy for respiratory issues are diarrhoea (Morrissey *et al.,* 2001).

C. crispus is harvested by plucking or with scissors, small knives, or hand-rakes from the shore (Mac Monagail and Morrison 2020). It may also be plucked with bare hands (Scottish Government *et al.,* 2016). Relative to England and Wales, *M. stellatus* and *C. crispus* are both harvested commercially to a larger extent in Scotland and Ireland, where the frond is removed but the base is left intact to allow re-growth.

C. crispus is also harvested commercially in Spain, France, Portugal, Canada and North America (Guiry & Blunden, 1991), and may be hand-raked, drag-raked from a boat, or gathered as storm cast from beaches, particularly in Canada where is forms a substantial component of drift weed, called “storm toss” (MacFarlane, 1968; Pringle, 1986). In Ireland, the seaweed industry has experienced a decline since its peak in the early 1960s (Pybus, 1977). In Ireland it is also harvested from shallow water by dredging with specialised rakes (Morrissey *et al.,* 2001)

Chondrus crispus: Natural range / distribution

C. crispus is found attached to rocks and boulders from the middle to lower rocky shore and in tide pools. It also occurs at depths of up to 24 m. It can tolerate some reduction in salinity and can be found in estuaries with salinities from 16-35 psu (Pybus, 1977; Mathieson & Burns, 1975). Found from sheltered to exposed shores, it can form extensive beds in areas with some wave shelter (Scrosati, 2016). It is typically found from moderately strong to strong water flow, but can tolerate a range of current flows, from negligible to tidal rapid of up to 5.5 knots.

Widely distributed on rocky shores on all British and Irish coasts (See Figure 3). Globally it occurs in Iceland, the Faroes, the western Baltic Sea, from northern Russia to southern Spain, the Mediterranean, Portugal, the Azores and West Africa. In north America it occurs in Alaska and from Labrador in Canada to New Jersey in the USA. Also occurs in the Bering Sea (East Asia).

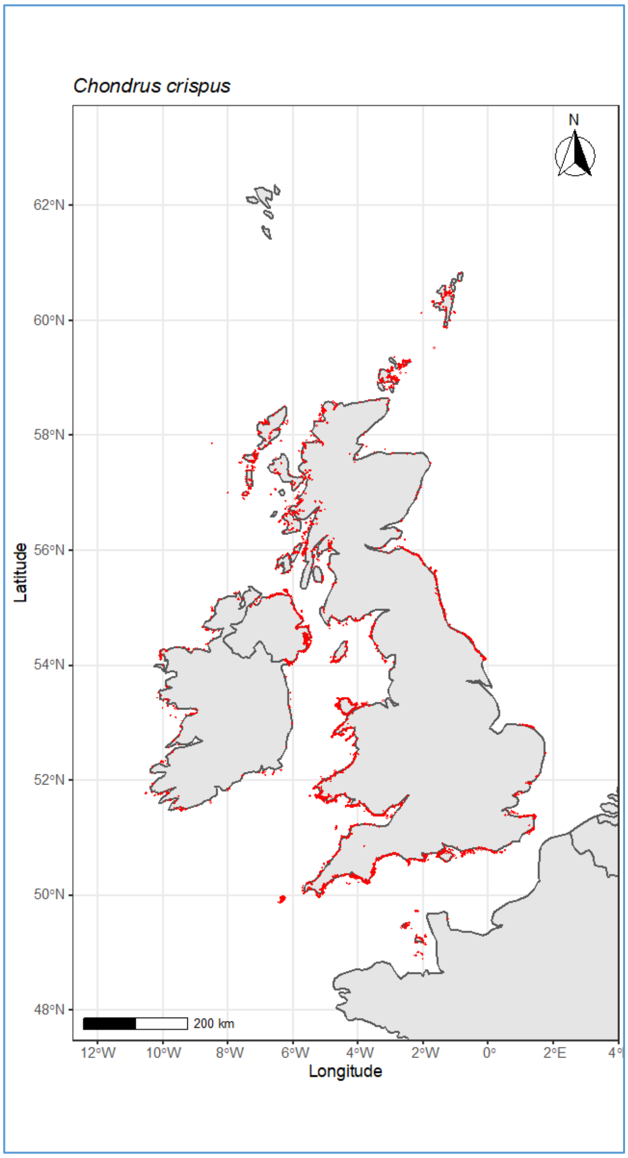

Figure 3. Distribution of *C. crispus* in the United Kingdom and Ireland. Occurrences of *C. crispus* plotted as red points. Spatial data of species occurrences obtained from NBN Atlas, DASSH and Marine Recorder databases.

Chondrus crispus: Growth, reproduction and seasonality

C. crispus grows up to 22 cm long. Pybus (1977) estimated that *C. crispus* reaches maturity approximately 2 years after the initiation of the basal disc, at which stage the fronds were approximately 12 cm long. The holdfasts of individual *C. crispus* can coalesce over time and can form an extensive crust on rock (Taylor *et al.,* 1981).

Longevity is typically 2-3 years (Taylor, cited in Pringle & Mathieson, 1986) but may live up to 6 years in sheltered waters (Harvey & McLachlan, 1973). The holdfast has greater longevity than the frond, and is capable of regenerating new fronds after damage or loss

(Mathieson & Burns, 1975; Dudgeon & Johnson, 1992; Taylor, cited in Pringle & Mathieson, 1986).

Growth rates vary widely according to environmental conditions. Peak growth occurs from May to November in eastern Canada (Juanes & McLachlan, 1992; Chopin *et al.,* 1999), although Pybus (1977) reported little seasonal variation in growth from populations in Galway Bay. Growth rates of 0.33 - 0.37 mm/day have been reported from the field (Pybus (1977; Prince & Kingsbury, 1973), and 0.02-0.08 mm/day in culture (Tasende & Fraga, 1999).

Growth rate appears to be temperature controlled (Tasende & Fraga, 1999), with optimum temperatures reported from laboratory conditions varying widely: 10-15°C (Fortes & Lüning, 1980), 15-17°C (Tasende & Fraga, 1999) and 20°C (Simpson & Shacklock, 1979; Kuebler & Dudgeon 1996). Growth in the lab was found to increase towards full salinity, and decline with high light intensities (> 20 µmol/m²/s) and short day length (photoperiods of 16:8 light: dark) (Tasende & Fraga, 1999).

Reproduction

The life cycle of red seaweeds is complex, involving three distinct morphological stages. Alternation occurs between asexual spore producing stages (tetrasporophytes) and male and female plants producing sexually. In *C. crispus* the tetrasporophyte phase appears as a flat, dark red - black crust, with a rubbery appearance. This is known as the Petrocelis and capable of growing laterally and covering extensive areas. The basal crusts and crustose tetrasporphytes are perennial, tough, resistant stages that may prevent other species from occupying the rock surface and allow rapid regeneration. They may therefore provide a significant recovery mechanism. The plants seen on the shore are gamtophyte stages, with separate male and female plants, which are distinguishable from one another only by microscopy.

Fruiting bodies are found on the upper parts of tips of the frond (Edwards *et al.,* 2012). Spore output of *C. crispus* has been reported to be similar for both gametophytes and tetrasporophytes, with an estimated number of spores being 8 x 1010/m²/year (Fernandez & Menendez, 1991).

Reproductive Season

Dickinson (1963) reported that *C. crispus* was fertile in the UK from autumn to spring, but that the exact timings varied according to local environment. Similarly, Pybus (1977) reported that although carposporic plants were present throughout the year in Galway Bay, Ireland, maximum reproduction occurred in the winter and estimated that settling of spores occurred between January and May.

In northern Spain, the species had reproductive capacity all year round but was greatest for gametophytes between November and March and for tetrasporophytes in April (Fernandez & Menendez, 1991). In Nova Scotia, Canada, cystocarps and tetrasporangia have been recorded on *C. crispus* all year round with a reproductive peak from August to October (Scrosati *et al.,* 1994). However, spores failed to germinate below 5°C and so winter temperatures in Nova Scotia are unsuitable for spore germination. Scrosati *et al.* (1994) also commented that viability of spores was low (<30%) and suggested that reproduction by spores probably does not contribute much to maintenance of the intertidal population of *C. crispus* in Nova Scotia, compared to vegetative growth of gametophytes.

Chondrus crispus: Recruitment and recovery potential

Life history summary

The life history characteristics of *C. crispus* give the species a strong capacity for recovery (although see below with regard to repeated heavy exploitation). It has an extended reproductive period (e.g. Pybus, 1977; Fernandez & Menendez, 1991; Scrosati *et al.,* 1994) and produces large numbers of spores (Fernandez & Menendez, 1991). Although growth is relatively slow compared to that of brown macroalgae, maturity is probably reached approximately 2 years after initiation of the basal disc (Pybus, 1977) and the fronds may persist for up to 6 years (Harvey & McLachlan, 1973). The basal crusts and crustose tetrasporphytes are perennial, tough, resistant stages that may prevent other species from occupying the rock surface and allow rapid regeneration. They may therefore provide a significant recovery mechanism.

Dispersal

The spores of red algae are entirely reliant on the hydrographic regime for dispersal (Norton, 1992). Recruitment of *C. crispus* is likely to occur on a very local scale, such as within 10 m of the parent plant. Recovery of a population of *C. crispus* is therefore likely to be largely dependent on whether holdfasts remain, from which new fronds can regenerate (Holt *et al.,* 1995).

Recovery from experimental harvesting

Following experimental harvesting by drag raking in New Hampshire, USA, populations recovered to 1/3 of their original biomass after 6 months and totally recovered after 12 months (Mathieson & Burns, 1975). Raking, often conducted from a small boat either side of low tide, is designed to remove the large fronds but leave the small upright shoots and holdfasts. The authors suggested that control levels of biomass and reproductive capacity are probably re-established after 18 months of regrowth.

Time to recovery appears to be highly seasonal, taking much longer after winter harvesting than in spring or summer (Mathieson & Burns, 1975).

Recovery following total clearance

Following total clearance of all algae by ice scour, recolonization by macroalgae occurred initially by fucoids and then perennial red seaweeds. After 2 years, *C. crispus* had re-established approximately 50% cover on the lower shore and after 5 years it was the dominant macroalga at this height, with approximately 100% cover, indicating that *C. crispus* was a succession climax species (Minchinton *et al.,* 1997).

In the short term therefore, harvesting of *C. crispus* may remove biomass and impair reproductive capacity, while in the long term, it has the potential to alter community structure and change the dominant species.

In summary, recovery by *C. crispus* will be relatively rapid following a single harvesting event in situations where some holdfasts remain for regeneration of fronds. In situations where the entire population is removed, recovery will be limited by recruitment from a remote population. If unregulated, long term heavy exploitation can severely deplete the seaweed resource.

Chondrus crispus: Ecological importance of species

Habitat provision

This species can form extensive stands in wave-sheltered locations where it acts as a foundation species, creating a habitat which can increase the abundance and species richness of invertebrate species and other algal species (Scrosati, 2016). A total of 69 species were found in association with *C. crispus* beds in Nova Scotia (Scrosati, 2016), The extent of influence as a foundations species is considered to be moderate, and *C. crispus* was found to be more strongly correlated with benthic species richness than fucoid cover on the same shore.

Habitat provision-understorey flora and fauna

It has been suggested that the majority of animals found within *C. crispus* beds are small (e.g. amphipods, bryozoans, crustaceans gastropods, nematodes, etc.) or juvenile stages of larger organisms like urchins, mussels, crabs and sea stars (Lilly, 1968 in FAO.org).

Mytilus edulis can settle in beds with detrimental effects, while grazing by *Littorina littorea,* green urchin *Strongylocentrotus droebachiensis*, *Haliotus* and *Acmea testudinalis* have been recorded (Lubchenco and Menge, 1978; MacFarlane, 1968).

Habitat provision-epiphytes

In Ireland, *C. crispus* have been found to host various species of algal epiphytes and fauna (Pybus, 1977).

The bryozoan *Alcyonidium polyoum* preferentially settles on *C. crispus* rather than other algae (Hurlbut, 1991). The seaweeds, *Ulva* sp. and *Ectocarpus* sp. grew epiphytically on *C. crispus* in culture, and were in turn grazed by the crustaceans *Gammarus lawrencianus* and *Idotea baltica* (Shacklock & Doyle, 1983).

Habitat modification

At high tide, dense stands of *C. crispus* mitigate water flow velocity reducing hydrodynamic stress for mobile invertebrates sheltering within the algal beds (Johnson 2001, Boller and Carrington 2006).

Chondrus crispus: Ecological implications of harvesting

Impact of harvesting on *C. crispus* beds can result in reduced reproductive output, and declines in abundance resulting in replacement by other seaweed species. Chopin *et al.*

(1988) and Sharp and Pringle (1990) both reported that harvested beds of *C. crispus* showed reduced reproductive capacity than unharvested beds.

Increased harvesting intensity has been reported to impact the size-class structure of the population by reducing mean frond length (McLaughlin *et al.*, 2006).

Habitat provision

Species which shelter within *C. crispus* beds are likely to either be removed or displaced to some extent by harvesting activity.

Changes in community composition

In Canada, prior to 1980, seaweed beds were dominated by *C. crispus* and the species was heavily exploited. However, the species declined in abundance and appears to have been replaced by another red seaweed, *Furcellaria lumbricalis,* which is thought to be the result of harvesting pressure (Sharp *et al.,* 1993). Similarly, Minchinton *et al.* (1997) and MacFarlane (1952) report that following destruction *C. crispus* beds initial recolonization was dominated by fucoids and the brown algae *Chordaria* respectively, with the perennial red seaweed taking 4-5 years to re-establish dominance. Clearly these difference in macroalgal canopy forming species will impact on biodiversity and ecological function.

Similarly MacFarlane (1952) reports that the particularly harsh winter of 1947/48 destroyed *C. crispus* beds near Pubnico. By the next summer, the annual brown alga *Chordaria* had colonized the area, and by the summer of 1950, *Fucus* had taken over as the dominant successional stage. *C. crispus* did not noticeably start to grow back in the area until summer 1951, four years later. Pringle and Semple (1980) estimated it would take about four years for a bare patch in a *C. crispus* bed to fill in with harvestable plants and five to ten years for *C. crispus* to re-establish in barren areas.

Turf forming biotopes and fucoid dominated biotopes may represent alternate stable states that continue while the dominant turf or fucoids occupy space. Removal of the turf may therefore allow re-establishment of a Fucoid or kelp dominated biotope that will remain until environmental or other factors again alter the state. Lubchenco (1980) for example, on shores in New England, found that the removal of *C. crispus* turf allowed the establishment of *Fucus* spp. MacFarlane (1952) also recorded a shift to a *Corallina officinalis* and encrusting coralline biotope following over raking (for harvesting) of *C. crispus* turf, in these areas gastropods had increased in abundance and prevented the recovery of *C. crispus* by grazing.

It should therefore be noted that where red algal turfs are removed, recovery may be prolonged. In Nova Scotia, high *C. crispus* cover was associated with a greater diversity of invertebrate species but fewer algal species than low *C. crispus* cover (Scrosati., 2016). Hence, the proportion of the canopy harvested is likely to influence the composition of the associated community.

Relevant studies or monitoring

C. crispus is a relatively well studied species (see examples above). Harvesting on a commercial scale has been best studied in Canada. Harvesting in the north west Atlantic is centred on the Gulfs of Maine and St Lawrence where the species is dominant has been well studied (Pringle & Mathieson, 1986). The annual catch peaked in 1974 at approximately 50,000 t and has since declined, due in part to decreased demand, but declines due to overexploitation have been implicated in a regime shift (Pringle & Mathieson, 1986).

Most of the evidence for recovery of *C. crispus* is based on experiments that simulate the effects of different harvesting mechanisms and intensities (Macfarlane, 1952; Mathieson & Burns, 1975).

Biomass assessments

Sharp *et al.* (1986) reported that the first drag-rake harvest of the season, removed 11% of *C. crispus* fronds and 40% of the biomass, but efficiency declined as the harvesting season progressed.

Standing stock biomass estimates exist for the species in Nova Scotia, although these are now rather dated.

Species specific management approaches

- Harvest a small proportion of the frond; leave the holdfast and some frond branches
- Harvest during spring/summer when peak growth occurs
- Avoid harvesting during reproductive / slow growth season in autumn/winter
- Ireland: harvest exposed sites every 2 years and sheltered sites every 4 years; leave mature plants to grow every 2 m (Edwards *et al.*, 2011).

In Ireland, harvesting has generally remained sustainable through pickers developing an intuitive feel for the annual cycle of local stocks and certain practices which involve pulling only the bushy top half of the frond off leaving the base and holdfast behind (Morrissey *et al.,* 2001). With favourable conditions, yield can be as much as 150 kg (wet weight) per spring tide (Morrissey *et al.,* 2001).

To summarise: In Canada the resource is managed by a government agency (see Pringle, 1986), which controls harvest rates by species, area, season, gear type, etc. along with regulation enforcement. Generally, resource management plans are modified annually by an advisory committee, by controlling both the annual number of harvesting days (seasons) and effort within marine plant harvesting districts. Harvesting gear type is also controlled based on impact to the target species and other commercially important species in the community. This resulted in a ban on use of the basket drag-rake in 1977. A minimum handrake tine spacing of 7.0 mm was introduced in one district in 1979. There is evidence from this district that annual yield would be enhanced by varying each season's opening date in order to better correspond with seasonal growth and tide regime.

Chondrus crispus: Knowledge Gaps

C. crispus is a relatively well studied, although most research has been conducted from Ireland and the western Atlantic.

- Standing stock biomass is unknown;
- Regional growth rates in England and Wales appear to be variable.

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Appendix 4 *Fucus serratus*

Names

Fucus serratus

Serrated wrack, Toothed wrack, Saw wrack, Irish: Mioranach, Dulaman, Murach Dhubh

Fucus serratus: harvesting

Uses of *F. serratus* include bodycare products, seaweed baths, and as an agricultural fertilizer. It has a high mineral and vitamin content with antioxidant properties, and extracted oils are used in cosmetics and health treatments (thalassotherapy). Bioactives from *F. serratus* are described as having antioxidant, antitumor, anti-venom and anticoagulant properties (Edwards *et al.,* 2013; Hoare *et al.,* 2019).

F. serratus is harvested throughout the UK at low tide.

Fucus serratus: Natural range / distribution

F. serratus grows in high densities on hard substrata (i.e. bedrock, cobbles) on the low shore in sheltered to moderately exposed areas of coastline (d'Avack & Garrard, 2015). Increases above this level of wave action will cause damage to individual plants, breaking fronds and removing entire plants from the substratum.

Propagule dispersal, fertilization, settlement, and recruitment are also influenced by water movement (Pearson & Brawley, 1996). In addition, increased water flow will cause scour, though increased sediment movement, affecting in particular small life stages of macroalgae by removing new recruits from the substratum and hence reducing successful recruitment (Devinny & Volse, 1978).

Growth rates for *F. serratus* are maximal at a salinity of 20 psu with the critical limit for recruitment set at 7 psu (Malm *et al.,* 2001). Sufficient salinity is essential for successful fertilization and germination in Fucus (e.g., Brawley, 1992; Serrão *et al.,* 1999; Malm *et al.,* 2001, Serrão *et al.,* 1996).

Several studies have observed adverse effects *F. serratus* as a result of thermal stress in terms of growth, physiological performance and reproductive output in Spain and Portugal (Pearson 2009; Viejo *et al.,* 2011; Martínez *et al.,* 2012, Jueterbock *et al.* 2014).

Found throughout the UK where suitable substrata exists (See Figure 4). It’s distribution ranges from Northern Portugal to Iceland and the Gulf of St. Lawrence in the western north Atlantic.

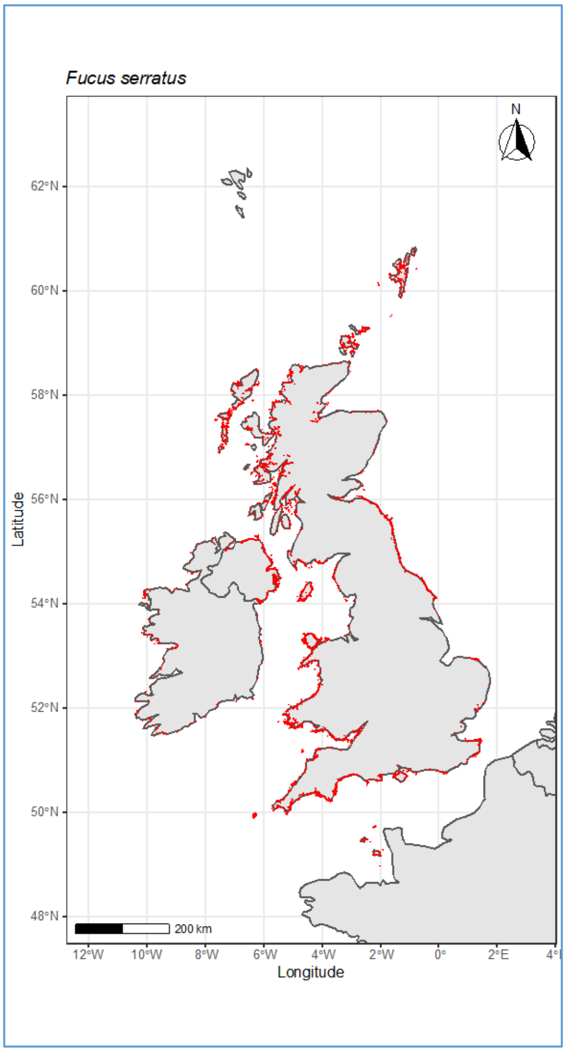

Figure 4. Distribution of *F. serratus* in the United Kingdom and Ireland. Occurrences of *F. serratus* plotted as red points. Spatial data of species occurrences obtained from NBN Atlas, DASSH and Marine Recorder databases.

Fucus serratus: Growth, reproduction and seasonality

F. serratus is a short-lived perennial, with separate male and female plants. Mature individuals typically grow up to 70 cm but have been recorded at over 2 m in length in very sheltered environments.

Growth rate generally ranges from 4-12 cm per annum, however, this can vary considerably depending on environmental conditions. During most of the year plant densities range between 10-14/0.25 square metres. During recruitment periods densities may rise to 18-22/0.25 square metres. Surface cover by this species may reach over 95 percent during the summer. This decreases and becomes patchier during winter and autumn as many plants may be lost due to storms and heavy wave action.

Sexual reproduction commences in late spring/early summer and continues through summer and autumn, peaking in August - October. Eggs and sperm are released into the water and fertilization occurs in the water column. The zygote then develops into a minute plant that can then settle onto the substratum. Arrontes (1993) determined that the dispersal of *F. serratus* gametes and fertilized eggs was restricted to within 1–2 m from the parent. Average annual expansion rates for *F. serratus* have been estimated at 0.3 to 0.6 km per year (Coyer *et al.,* 2006; Brawley *et al.,* 2009). Dispersal is highly limited as the negatively buoyant eggs are fertilized almost immediately after release and dispersal by rafting reproductive individuals is unlikely (Coyer *et al.,* 2006).

The number of reproductive receptacles increases with age, so large mature plants contribute a disproportionately large amount to reproductive output.

F. serratus does not float, and thus mature detached individuals cannot transport reproductive material to distant sites as might be the case for other brown algae. However, *F. serratus* is found on all British and Irish coasts so there are few mechanisms isolating populations. While poor dispersal is true for medium or large spatial scales (hundreds of metres to kilometres), recruitment at short distances from parental patches is very efficient, as most propagules settle in the vicinity of parent plants (Arrontes, 2002).

Temperature effects on growth

Most fucoids are cold-temperate species (Lüning, 1984). *F. serratus* shows optimal growth at 20 °C (Jackson, 2008) and so is well within its thermal range in the British Isles. Although, under laboratory conditions*, F. serratus* has shown no decline in growth at 22 °C (Nielsen *et al.,* 2014) or 25 °C (Arrontes,1993). Temperatures above the thermal maxima of 20°C are generally considered unsuitable for these algae (Zou *et al.,* 2012), with an inability to survive continual exposure to temperatures above 28°C (d'Avack & Garrard, 2015). Growth rates of adult brown macroalgae may be affected by temperature through the increase in metabolic rates (Nygard & Dring, 2008). Germlings are also negatively affected by increased temperature (Neilsen *et al.,* 2014). This indicates that early life stages could be more vulnerable than mature algae to this pressure (Neilsen *et al.,* 2014).

Fucus serratus: Recruitment and recovery potential

No evidence was found of the effect of hand harvesting *F. serratus* following Welsh and English guidance (i.e. leaving the lower part of each plant). Provided the plants are not removed entirely, regeneration from the remaining fronds is likely.

Growth varies with environmental conditions, but during peak growth in summer, the canopy may be expected to recover within a matter of months. Removal of some of the adult canopy will allow more light to the understorey, so germlings and juveniles will be able to grow more quickly (d'Avack & Garrard, 2015).

Following total clearance, rapid recolonization is possible if some adults remain nearby due to high fecundity and efficient recruitment over short distances, although if the entire population of *F. serratus* is lost other species may come to dominate (Schiel & Foster (2006).

After experimental (small scale 2 sq. metre) canopy removal of *F. serratus* on a moderately exposed shore, the *F. serratus* cover recovered within one year (Hawkins & Harkin, 1985).

Dispersal will be poor over medium or large spatial scales (hundreds of metres to kilometres), with recruitment at short distances from parental patches is very efficient, as most propagules settle in the vicinity of parent plants (Arrontes, 2002; Coyer *et al.,* 2006).

Fucus serratus: Ecological importance

Primary production and nutrient cycling

Due to relatively fast growth and productivity this species is likely to play an important role in the turnover of organic matter, export of detritus, carbon capture, and nutrient cycling however no literature specific to *F. serratus* was found by the rapid evidence assessment.

Habitat provision and modification

F. serratus provides habitat and food for a diverse community of species (Fredriksen *et al.,* 2005). As ecosystem engineer's fucoid algal canopies modify habitat conditions, facilitating the existence and survival of other intertidal species and therefore strongly influence the structure and functioning of coastal ecosystems (Jenkins *et al.,* 2008). Associated fauna of three biotope variants have been described: *F. serratus* with red seaweeds (Fser.R) and *F. serratus* with under-boulder communities (Fser.Bo) with sponges, and *F. serratus* and piddocks community on soft rock. Dense *F. serratus* with fewer red seaweeds occurs on more sheltered shores (Fserr) (Connor *et al.,* 2004; JNCC, 2015).

Fucus serratus: Ecological implications of harvesting

Primary and secondary production

Harvesting removes seaweed canopies which will have important direct and indirect effects on the wider ecosystem. Stagnol *et al.* (2013) investigated the effects of commercial harvesting of intertidal *F. serratus* on ecosystem biodiversity and functioning. The study found that the removal of macroalgae affected the metabolic flux of the area. Flows from primary production and community respiration were lower on the impacted area as the removal of the canopy caused changes in temperature and humidity conditions. Suspension feeders were the most affected by the canopy removal as canopy-forming algae are crucial habitats for these species, most of them being sessile organisms.

Habitat provision and modification

The loss of *F. serratus* canopy will have both short and long-term consequences for associated benthic communities, resulting in the loss of biogenic habitat, reduction in diversity, simplification of vertical structure and reduction or loss of ecosystem functioning such as primary productivity (Lilley & Schiel, 2006). The removal of macroalgae canopy exposes understorey species to sunlight, desiccation and aerial conditions resulting in bleaching and eventual die backs (d'Avack & Garrard, 2015).

Stagnol *et al.* (2013) investigated the effects of commercial harvesting of intertidal *F. serratus* finding that canopy loss seemed to have a negative impact mainly on the diversity of the animal community.

Changes in community composition

Re-establishment of *F. serratus* may depend on the ability to out-compete other species and this may be dependent on suitable environmental conditions. Indeed, loss of fucoids can cause systems shifts to a state dominated by low-lying turf or filamentous ephemeral algae (Airoldi *et al.,* 2008; Mangialajo *et al.,* 2008; Perkol-Finkel & Airoldi, 2010; Schiel & Foster 2006).

Turf algae, especially corallines, are often highly resilient and positively associated with perturbed areas, and can recover and reach greater abundance compared to prior disturbance conditions (Bulleri *et al.,* 2002; Bertocci *et al.,* 2010). These turf algae can then prevent canopy recovery by inhibiting recruitment.

Stagnol *et al.* (2013) observed *Patella vulgata* recruiting in bare patches of disturbed plots. Experimental studies have shown that limpets control the development of macroalgae by consuming microscopic phases (Jenkins *et al.,* 2005) or the adult stages (Davies *et al.,* 2007). The increase in *Patella vulgata* abundance could thus limit the recruitment and growth of *F. serratus* on the impact zone. Stagnol *et al.* (2013) found that opportunistic ephemeral green algae such as Ulva sp. responded positively to removal of the *Fucus* canopy. These green ephemeral algae are major competitors of *F. serratus* for space colonization and nutrient uptake. Blooms of ephemeral algae facilitated by disturbance may then slow the development of longer-lived perennial algae, especially fucoids.

Relevant studies or monitoring

In Brittany, Stagnol *et al.* (2013) investigated the effects of commercial harvesting of intertidal *F. serratus* on ecosystem biodiversity and functioning, although all biomass was cleared from 25 m^2 areas, which is not representative of harvesting following guidance in England and Wales.

Species specific management approaches

- Cut fronds at a certain height above the base
- Avoid harvesting during the reproductive season (autumn – winter)
- Avoid harvesting only large mature individuals, which contribute a disproportionately large amount to reproduction.

In Scotland, government guidance advises *F. serratus* to be cut 15-25 cm above the holdfast (Scottish Government *et al.,* 2016). In England and Wales, it is advised to cut 30cm above the holdfast and to avoid harvesting during the spring/summer reproductive period (Bailey & Owen 2014; NRW, 2018).

Knowledge Gaps

- Standing stock biomass in England and Wales
- Sustainable harvesting volumes

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Appendix 5 *Fucus vesiculosus*

Names

Fucus vesiculosus

Bladder wrack, Black tang, Bladder fucus, Dyers fucus, Red fucus, Rock wrack

Fucus vesiculosus: harvesting

F. vesiculosus has a high mineral and vitamin content with antioxidant properties and bioactives that have the potential to ameliorate body tissue (Hoare *et al.,* 2019). Extracted oils are used in cosmetics and health treatments (thalassotherapy), but also in agriculture as a liquid fertiliser. Recently*, F.vesisculosus* was found to have antimicrobial activity against bacterial infections (i.e. MRSA), raising potential use in therapeutics (Hoare *et al.,* 2019).

F. vesiculosus is harvested throughout the UK, typically using scissors or small knives at low tide.

Fucus vesiculosus: Natural range / distribution

Distribution ranges across the Atlantic, from the west coast of Portugal and eastern USA to Northern Norway. *F. vesiculosus* can survive within a broad temperature range of 10-24 °C (Graiff *et al.,* 2015). However, growth rate and survivorship can strongly decline above 20°C, becoming lethal at 28°C if exposed for 1 week (Lüning, 1990). In the UK, *F. vesiculosus* is found in the middle of its natural temperature range (See Figure 5) and will therefore not be affected by an increase in 5 °C for one month or an increase of 2 °C for one year. However, extended periods of increasing temperatures will likely negatively affect growth rates and therefore canopy cover of *F. vesiculosus* (Khan *et al.,* 2017). Under projected climate change, the disappearance of *F. vesiculosus* is predicted by 2200 from North Atlantic shores south of 45° latitude (Jueterbock *et al.,* 2013)

F. vesiculosus grows in high densities on hard substrata (i.e. bedrock, cobbles, pebbles) on the low shore in sheltered areas of coastline (d'Avack & Garrard, 2015). In some sheltered areas *F. vesiculosus* forms a narrow zone above the *Ascophyllum nodosum* zone (Connor *et al*., 2004; JNCC, 2015).

As an intertidal species with a wide latitudinal distribution, *F. vesiculosus* is well adapted to cope with varying salinities over temporal and spatial scales. In the Baltic Sea, *F. vesiculosus* is adapted to grown in salinities down to 4 psu (Kautsky, 1992). Bäck *et al.* (1992) compared *F. vesiculosus* individuals from Atlantic and the Baltic populations. Both populations were able to withstand a wide range of salinities in laboratory cultures, yet some differences were recorded. The Atlantic population showed better growth in higher salinities and virtually no growth at 5 ppt. Those individuals kept at 5 ppt mortality occurred after 7 weeks. In contrast, the Baltic wracks grew better in conditions with lower salinities. Growth was negligible at the highest tested salinity (45 ppt). Back *et al.* (1992)

demonstrates that sensitivity of *F. vesiculosus* to changes in salinity differ between populations.

F. vesiculosus can tolerate desiccation until the water content is reduced to ~ 30% (d'Avack & Garrard, 2015). If desiccation occurs beyond this level, irreversible damage occurs. Individuals at the top of the shore live at the upper limit of their physiological tolerance and are therefore likely to be unable to tolerate increased desiccation and would be displaced by more physiologically tolerant species. Tolerance varies between population, Gylle *et al.* (2009) found that *F. vesiculosus* populations naturally occurring in fully saline conditions had a higher emersion stress tolerance compared to brackish populations. Early life history stages are more susceptible to this pressure compared to adults (Henry & Van Alstyne, 2004). Germlings are however protected from desiccation by the canopy of adults. Desiccation and the associated osmotic stress, especially when combined with high temperatures can cause mortalities (Pearson *et al.,* 2009). Jonsson *et al*. (2006) found that flow speed of 7-8 m/s completely dislodged *F. vesiculosus* individuals larger than 10 cm. Slower water movement can also cause oxygen deficiency directly impacting the fitness of algae (Olsenz, 2011).

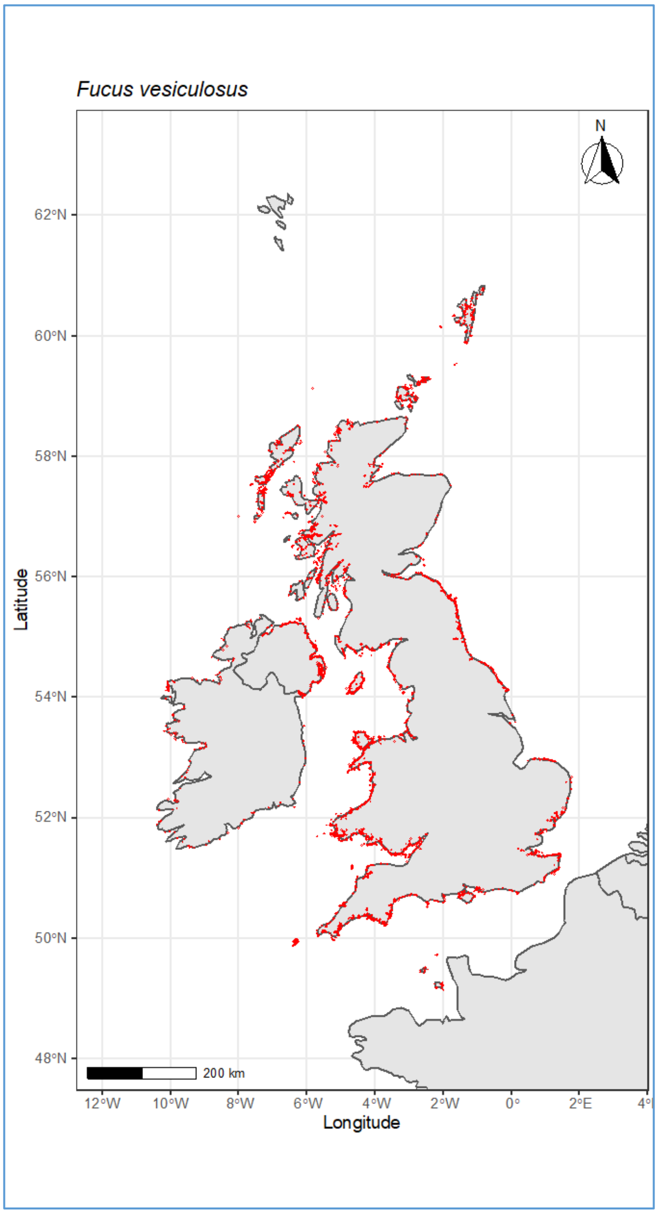

Figure 5. Distribution of *F. vesiculosus* in the United Kingdom and Ireland. Occurrences of *F. vesiculosus* plotted as red points. Spatial data of species occurrences obtained from NBN Atlas, DASSH and Marine Recorder databases.

Fucus vesiculosus: Growth, reproduction and seasonality

F. vesiculosus is a short lived perennial, reaching a maximum of about 4-5 years old (S. Kraan, pers. comm.). In addition to reproducing sexually, it has the ability to generate vegetative regrowth in response to damage. Under sheltered conditions, the fronds have been known to grow up to 2 m (Wippelhauser, 1996).

F. vesiculosus is a fast growing species, able to colonize patches of clear substratum rapidly. Relative growth rate can vary from 0.05-0.14 cm/day depending on temperature and light conditions (S. Kraan, pers. comm.). The increase in growth rate for *F. vesiculosus* at 10, 12.5 and 15 °C was found to be, on average, 28% higher than it was at 7 °C

(Strömgren, 1977). In the northern Baltic, the highest relative growth rate of vegetative branches for *F. vesiculosus* was observed in the summer (up to 0.7% / day) compared to winter growth (less than 0.3% / day). In Sweden, growth rates of 0.7-0.8 cm / week were reported over the summer months of June and August (Carlson, 1991).

Growth rate can also vary with exposure. In an exposed site in Scotland, *F. vesiculosus*, grew about 0.31 cm / week whereas plants at more sheltered site grew an average of 0.68 cm / week (Knight & Parke, 1950). The proportion of energy allocated between vegetative and reproductive growth also varies throughout the year. In the northern Baltic, reproductive branches experienced a peak in growth rate in mid April where the relative growth rate was almost 0.1% / day (Lehvo *et al.,* 2001).

Reproduction

Development of the receptacles takes approximately three months from initiation until when gametes are released (Knight, 1947). On British shores, receptacles are initiated around December and may be present until late summer (Knight, 1947), with a peak of fertility in May and June. Males and females occur as separate plants. Eggs and sperm are generally released into seawater under calm conditions whereby external fertilisation occurs (Mann, 1972; Serrão *et al.,* 2000). Serrão *et al.* (1997) determined that the wrack had a short-range dispersal capacity. Under calm conditions in which eggs are released, most eggs fall in the immediate vicinity of the parent plants. The egg becomes attached to the rock within a few hours of settlement and adhere firmly enough to resist removal by the next returning tide and germling may be visible to the naked eye within a couple of weeks (Knight & Parke, 1950).

Despite the poor long range dispersal, the species is highly fecund often bearing more than 1000 receptacles on each plant, which may produce in excess of one million eggs. On the coast of Maine, sampling on three separate occasions during the reproductive season revealed 100% fertilization on both exposed and sheltered shores (Serrão *et al.*, 2000). Fertilization is thus not considered as a limiting factor in reproduction in this species (Serrão *et al*., 2000). However, mortality is extremely high in the early life stages due to increased susceptibility environmantal stressor (Steen, 2004; Fredersdorf *et al.,* 2009), and biotic interactions such as predation (Knight & Parke 1950).

The number of reproductive receptacles increases with age, so large mature plants contribute a disproportionately large amount to reproductive output.

Fucus vesiculosus: Recruitment and recovery potential

McCook & Chapman (1992) experimentally damaged *F. vesiculosus* holdfasts to test the ability of the wrack to regenerate. The study found that vegetative sprouting of *F. vesiculosus* holdfasts made a significant addition to the regrowth of the canopy, even when holdfasts were cut to less than 2 mm tissue thickness. Four months after cutting, sprouts ranged from microscopic buds to shoots about 10 cm long with mature shoots widespread after 12 months. Vegetative regrowth in response to wounding has been suggested as an important mean of recovery from population losses (McLachan & Chen, 1972). The importance of regeneration will depend on the severity of damage, not only in terms of the individuals but also in terms of the scale of canopy removal (McLachan & Chen, 1972).

F. vesiculosus recruits rapidly to cleared areas of the shore and full recovery takes 1-3 years in British waters (Hartnoll & Hawkins, 1985). Keser & Larson (1984) investigated the recovery of *F. vesiculosus* to plots which had been scraped clean and burned with a propane torch. *F. vesiculosus* was the first perennial alga to colonize the experimentally denuded transects, even at sites and tidal levels that had been dominated by *Ascophyllum* or *Chondrus crispus* beforehand. Recovery occurred at all sites between 3 to 21 months. The study found newly settled germlings of *F. vesiculosus* in most months, indicating a broad period of reproduction. When grazers are excluded from areas of intertidal shores fucoids have the ability to rapidly recolonize areas, they can even be found in areas, which in a balanced ecosystem, they do not normally occur (Burrows & Lodge, 1950, Southward & Southward, 1978). Fucoid distributions return to their recognized zones when grazers are re-established on a shore (Burrows & Lodge, 1950, Southward & Southward, 1978). Although intertidal shores can rapidly regain fucoids, it can take considerably longer for ecosystem function to return if grazers have also been lost (Hawkins & Southward, 1992). If the whole community is removed, recovery is likely to occur at a much lower pace. Hawkins & Southward (1992) found that, after the M.V. Torrey Canyon oil spill, it took between 10 and 15 years for the *Fucus* spp. to return to 'normal' levels of spatial and variation in cover on moderately exposed shores. Therefore, for factors which are likely to totally destroy the biotope, recoverability is likely to be low.

Genetic diversity can influence the resilience of fucoids in particular when pressure persists over a long period of time. Genetically diverse populations are generally more resilient to changes in environmental conditions compared to genetically conserved populations. Tatarenkov *et al.* (2007) determined a high level of genetic variation in *F. vesiculosus* and extensive phenotypic variation. They suggested this might explain why the species is more successful than most fucoid species in colonizing marginal marine environments such as low-salinity estuaries, showing a range of morphological, physiological and ecological adaptations (Tatarenkov *et al.,* 2005). Pressures causing a rapid change will have a greater impact as the natural ability of the species to adapt is compromised.

Fucus serratus: Ecological importance of species

Habitat provision

F. vesiculosus provide substratum and shelter for a very wide variety of species such as, *Patella vulgata, Lacuna pallidula, Littorina mariae, Amphithoe rubricata, Idotea granulosa, Semibalanus balanoides, Nucella lapillus, Actinia equine, and Halichondria panacea*, to name a few (d'Avack & Garrard, 2015).

Fucus vesiculosus: Ecological implications of harvesting

Primary production and nutrient cycling

Stagnol *et al.* (2013) investigated the effects of commercial harvesting of intertidal fucoids on ecosystem biodiversity and functioning. The study found that the removal of the macroalgae canopy affected the metabolic flux of the area. Flows from primary production and community respiration were lower on the impacted area as the removal of the canopy caused changes in temperature and humidity conditions. Suspension feeders were the most affected by the canopy removal as canopy-forming algae are crucial habitats for these species, most of them being sessile organisms.

Changes in community composition

Other studies confirm that loss of canopy had both short and long-term consequences for benthic communities in terms of diversity resulting in shifts in community composition and a loss of ecosystem functioning such as primary productivity (Lilley & Schiel, 2006; Gollety *et al.,* 2008).

Relevant studies or monitoring

See above.

Species specific management approaches

- Cut fronds a certain height above the holdfast
- Avoid harvesting large mature individuals
- Avoid harvesting during the reproductive period

In Scotland, government guidance advise *F. vesiculosus* to be cut 15-25 cm above the holdfast (Scottish Government, 2016). In England and Wales, it is advised to cut 30cm above the holdfast, avoid harvesting large mature individuals (as these contribute more to reproduction), and to avoid harvesting during the spring and summer reproductive period (NRW, 2018).

Fucus vesiculosus: Knowledge Gaps

- Standing stock biomass
- Sustainable harvest volumes

Fucus vesiculosus: References

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Appendix 6 *Himanthalia elongata*

Names

Himanthalia elongata

Sea spaghetti, Thongweed, Sea bean, Irish: Raif, Riseach, Rualach

Himanthalia elongata: harvesting

H. elongata is eaten as Sea Spaghetti in the UK and Europe. The majority of harvesting is thought to take place from the shore, but also known to be collected by free diving in England. It is also harvested in Norway, France (particularly Brittany, in large quantities), Spain, Scotland and Ireland, for use as a fertilizer, human food, alginate extraction, pharmaceuticals (due to antioxidant, antibacterial and anti-UV properties) and beauty products such as facial cleansers.

H. elongata is commercially harvested year-round, but is most palatable in early summer, so may be more heavily targeted for domestic use in those months (Angus 2017).

Himanthalia elongata: Natural range / distribution

H. elongata is found attached to hard substrata such as bedrock and large boulders on the lower shore, where it forms a band below *Fucus serratus* and above the kelps.

The distribution of *H. elongata* appears to be controlled by the degree of wave exposure, presence of tidal currents and the availability of suitable substrata. The species grows best in areas with strong tidal currents (3 to 6 knots or 1.5-3 m/sec.) and is most commonly found on semi-exposed shores where it can be locally abundant. It is rarely found in exposed shores and occasionally forms dense stands on sheltered shores and can tolerate weak (< 1 knot or <0.5 m/sec.) current regimes.

This species is intolerant of lowered salinity and silt, restricting its distribution. *H. elongata* can sometimes extends into the shallow sublittoral and can tolerate the physical conditions found further up the shore, but it is prevented from growing there by grazing pressure.

Plants from exposed sites tend to be shorter and have fewer, narrower receptacles. Plants grow well in the presence of a red algal turf, which offers protection to young vegetative stages from wave action (Stengel *et al.,* 1999).

Vegetative growth in *H. elongata* peaks at temperatures 10-12 °C (Stengel *et al.,* 1999). This species appears to be tolerant of increases in temperature as plants survived the unusually hot summer of 1983 apart from a slight bleaching of buttons (Hawkins & Hartnoll, 1985).

The species is found from all coasts of Britain and Ireland, except southeast England (See Figure 6). It is also recorded from Norway, Britain, Ireland, northwest coast of France, northern Spain and Portugal.

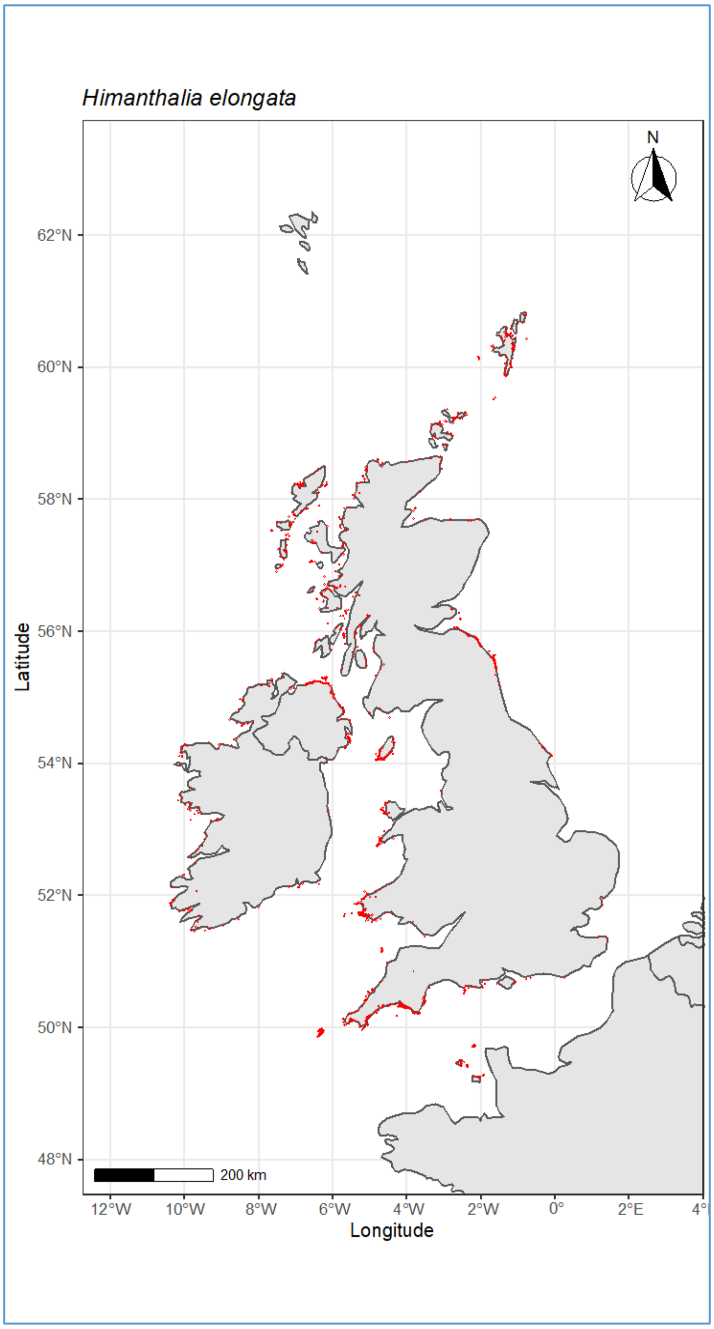

Figure 6. Distribution of *H. elongata* in the United Kingdom and Ireland. Occurrences of *H. elongata* plotted as red points. Spatial data of species occurrences obtained from NBN Atlas, DASSH and Marine Recorder databases.

Himanthalia elongata: Growth and reproduction and seasonality

H. elongata has a unique life history and growth pattern involving a two-stage morphology. Small button-like thalli are first produced, from which long strap-like reproductive fronds (receptacles) are formed in autumn. The button stage is clubbed shaped at first and then develops into a button shape 2-3 cm in diameter, which is connected to the substrate by a holdfast and short stipe. From the centre of each button typically sprouts 2 strap-like reproductive fronds in autumn, although plants have been observed with 1 to 4 straps. The strap-like reproductive fronds grow quickly between February and May, reaching a length of up to 2 m. This represents an investment of 98 percent of the total biomass in reproductive rather than vegetative tissue.

Male and females occur as separate plants. When the plants are fertile the straps become mottled with brown spots, each spot with a pale centre marking the opening to the conceptacle. The plant releases gametes at low tide by liberating them into mucus, which dribbles onto the substratum below. Gamete release occurs from June until the winter and plants commonly live for 2-3 years, reproducing once before dying.

Usually, germlings become visible on the shore in early March and form buttons with an average size of 10-25 mm by August. Those buttons which grow to 15 mm by November and produce receptacles that autumn, so age at maturity is between 9–14 months old. The receptacles grow little in length during autumn and winter but increase rapidly between February and May before releasing gametes from June until they decay over winter.

The time of reproduction is strongly site dependent, probably due to water temperature.

Zygotes of the species are very large in comparison to most seaweeds, and they rapidly settle to the substratum. After fertilisation there is a long period of 5-7 days before they develop attachment to the seabed. Gamete dispersal is thought to be limited so recruitment from external populations is probably low. Early germling growth is probably strongly influenced by the presence of adults, as reproductive thalli provide protection from desiccation and high irradiances, although shading could limit growth rate of germlings (Stengel, pers. comm. in White 2008).

Maximum growth rate of the reproductive straps is 16mm/day, occurring at 10-12 degrees °C during the spring. From May onwards growth slows, with maturation of receptacles aged 7–9 months in June (Stengel *et al.,* 1999).

The degree of exposure to waves is also important in determining the persistence of adult plants on the shore. On exposed sites, adult plants may only be present until October, whereas on sheltered sites, plants may be present until February.

Himanthalia elongata: Recruitment and recovery potential

Due to the large size and rapid settlement of zygotes, gamete dispersal is thought to be limited so recruitment from external populations is probably low (Moss *et al.,* 1973). However, *H. elongata* can rapidly recruit to cleared areas of the shore provided that an existing population is nearby (Stengel *et al.,* 1999), so as long as not too much of the population is harvested the species would be able to recover relatively quickly. Therefore, in close proximity to source populations (and depending on the time of year) recovery may take 1-2 years, however by recruitment from other populations may take longer.

Following recruitment, survival of early germlings is dependent on the presence of adult thalli or other foliose algae providing protection from desiccation, wave action and high irradiances, although shading probably limits growth rates of the germlings (Stengel *et al.,* 1999).

Recovery is thought to be quicker in the presence of grazers, which reduce the competition pressure of fast growing ephemeral algae (Aquilino and Stachowicz, 2012).

Himanthalia elongata: Ecological importance of species

Habitat provision- epiphytes

Kitching (1987), recorded 105 species of epiflora and epifauna on *Himanthalia* plants in Lough Hyne. The upper surface of the buttons are very resistant to colonization by epiphytes while the underside of the buttons are usually colonized by Bryozoa and spirorbid worms. The straps are frequently covered in a microforest of filamentous algae, such as *Ectocarpus siliculosus, Ceramium pedicellatum* and *Ulva prolifera*. Fauna such as the gastropods *Rissoa pavida* & juvenile *Patella pellucida* and amphipods, are often associated with the above filamentous epiphytes.

Habitat modification (including shading and moisture retention)

Germlings of *Himanthalia* appear to suppress germlings of *Fucus serratus*, as the 'mushroom' stage forms a canopy which overshadows *F. serratus* recruits, whereas at the slightly larger juvenile stage the upwards growth of *F. serratus* plants overshadows *H. elongata* 'buttons' (Choi *et al.,* 2005).

Himanthalia elongata: Ecological implications of harvesting

Habitat provision

A study in Brittany found that harvesting of *H. elongata* reduced its percentage cover for up to nine months, with full recovery apparent within 12 months. Harvesting had little impact on the associated community (Stagnol *et al.,* 2015), and removal of the canopy encouraged development of germlings. During the regrowth and recovery period, the ecological function of habitat provision will be absent or reduced. These impacts can be minimised when management guidance (removal of only one frond per button) is followed.

Relevant studies or monitoring

The species recruited to concrete blocks placed within an existent population at an average level of 45 buttons per block (968 sq. cm in area) in March of the first year, dropping to only 4 or 5 buttons per block by early summer, but rising to 1500 buttons per block by March of the second year (Stengel *et al.,* 1999).

Stagnol *et al.* (2015; 2016) experimentally harvested *H. elongata* and monitored recovery for 12 months, finding that removal of the canopy facilitates development of juvenile life stages, allowing for full recovery within 12 months.

In some areas of France and Portugal, harvesting (which is unregulated) increased 35% between 2009 and 2013 (Mac Monagail *et al.,* 2017), resulting in decreases in abundance of *H. elongata.* Harvesting for personal consumption in Portugal is unregulated and unmanaged, and *H. elongata* populations are locally in decline, with some local extinctions (Lima *et al.,* 2007) although it is unclear whether these are driven by climatic factors, competition, or harvesting activity.

Species specific management approaches

- Only harvest one of the two fronds per button
- Avoid harvesting fertile individuals / harvest after reproduction
- In Brittany, minimum harvesting sizes are legally enforced

Himanthalia elongata: Knowledge Gaps

Very limited evidence was found on sustainable harvesting levels, with no examples from England and Wales.

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Appendix 7 *Laminaria digitata*

Names

Laminaria digitata

Kombu, Tangle, Oarweed, Red ware, Sea girdle. Irish: Leath, Learach, Coirleach.

Known harvesting

Hand harvesting activity for *L. digitata* is relatively small scale in Wales and England, involving knives or scissors to harvest the blade both commercially and recreationally, primarily for human food use. It is reported to store sodium glutamate, so is marketed for its "Umami" flavour. In Scotland, it is harvested from small boats usually by stepping out of the boat at low tide and cutting the seaweed with a knife (Scottish Government, 2016). Juvenile plants are generally avoided.

L. digitata is also harvested (mechanically) on a large scale in Brittany for alginate.

The species probably contributes to a reasonably large extent to the volume of beach-cast kelp washed up on the shore, which is gathered for use as a soil conditioner or agricultural fertiliser. Collection of drift weed is seasonal following winter storms, and is not as widely reported from Wales and England as in Ireland or Scotland, where it is traditionally spread on machair land (Angus, 2017).

Laminaria digitata: Natural range / distribution

L. digitata is found at the lower intertidal sublittoral fringe where it grows attached to bedrock or other suitable hard substrata. It also occurs in deep rockpools up to mid-tide level. Where the two species co-occur, the lower limit of *L. digitata* is restricted by the presence of *L. hyperborea,* which outcompetes *L. digitata* (Kain, 1975). It flourishes in moderately exposed areas, but can tolerate sheltered to very exposed wave energy sites. Prefers strong water currents but tolerant of weak (< 1 knot or <0.5 m/sec.) to very strong (> 6 knots (>3 m/sec.). Usually found only in full salinity conditions, although tolerant of some seasonal reduction in salinity (Sundene, 1964).

Found throughout Britain where the habitat is suitable. Absent from Liverpool Bay and Severn estuary due to turbidity, and scarce on the south-east coast of England, due to lack of hard substrata (See Figure 7).

Also found in southern Greenland and east coast Canada, Quebec and North America from Hudson Straits to New York.

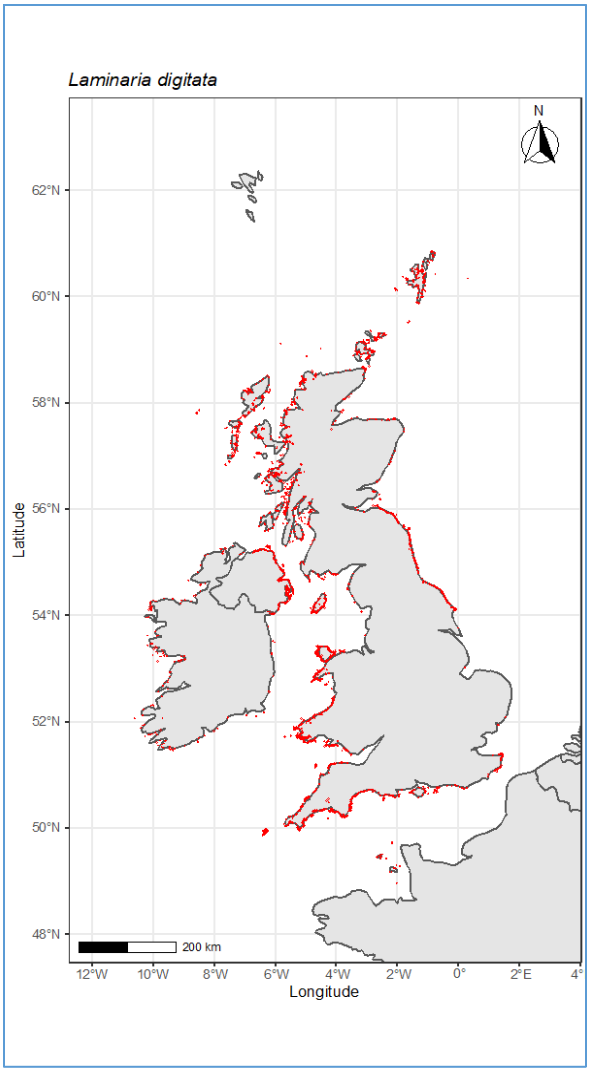

Figure 7. Distribution of *L. digitata* in the United Kingdom and Ireland. Occurrences of *L. digitata* plotted as red points. Spatial data of species occurrences obtained from NBN Atlas, DASSH and Marine Recorder databases.

Laminaria digitata: Growth, reproduction and seasonality

Reaches a maximum length of 2-4m, although 1-3m is more common, and an age of 4-6 years (Gatral & Cosson, 1973; cited in Birkett *et al.*, 1998b). Age at first maturity is 18-20 months.

Growth of the sporophyte is seasonal, with rapid growth from February to July followed by a period of slower growth from August to January. A mean growth rate of 1.3 cm / day has been reported (Pérez, 1971; cited in Kain, 1979), with new growth forming from the base of the blade (Dickinson, 1963).

Laminarian kelp all exhibit a two stage lifecycle, alternating between a visible sporophyte (the harvestable “plant”), which is usually of considerable size and a microscopic, filamentous gametophyte stage with separate male and female individuals. Optimal conditions prompt fertility; then the male gametes fertilise the female, and a juvenile sporophyte develops *in situ*.

Fertile “sorus” material is formed over most of the blade, from which large numbers (>1,000,000) of zoospores are released. Fertile material may be found year round, with peaks in July - August and November - December. The appearance of juvenile plantlets/germlings may also be year round, peaking in spring and autumn.

The microscopic gametophyte life stages typically last for ten days, however if conditions for germination are not favourable then fertility can be delayed, allowing gametophytes to continue to grow vegetatively until optimal conditions return (Chapman 1987). These gametophyte stages can serve as invisible ‘seed banks’, from which visible plants can develop, eventually appearing as small subcanopy plants in the understory.

The chemical composition of seaweeds varies seasonally, so the requirement for end use (i.e. nutrition, flavour, concentration of bioactives or carbohydrates) balanced with the need for maximum yield will influence harvest time (Manns *et al.,* 2017; Schiener *et al.,* 2015).

Laminaria digitata: Recruitment and recovery potential

Plants are able to rapidly re-colonize gaps on the shore which result from storm damage (Birkett *et al*., 1998) and after plant cutting the standing crop of *L. digitata* was re-established within 18-20 months (Kain, 1979).

In macroalgae clearance experiments at Port Erin, Isle of Man (Kain, 1975) recolonization of *L. digitata* on concrete blocks had taken place within 2 years. In France, Ciam (le Comité interprofessionel des algues marines) proposed that, regardless of collection method, the restoration of stands of Laminariales took up to 18 months after harvesting (Arzel, 1998).

Recovery of cleared plots in Helgoland to original density took longer, 25 months, probably because plots were burned to ensure all spores, gemetophytes and germlings were also removed (Markham & Munda, 1980). However, although the density of algal cover had returned to pre-clearance levels the *L. digitata* plants were smaller than those on undisturbed plots. This suggests full population recovery is longer than 25 months.

Season

Clearance experiments in Brittany observed recovery was more rapid following clearance in autumn than spring, but still took 12 months (Engelen *et al.,* 2011)

Dispersal potential

L. digitata spores may disperse at least 200 m, and up to 600 m away from the parent (Birkett *et al*., 1998; Chapman, 1981), settling on to available hard substrate on the seabed after 24 hours. For fertilisation to be successful, gametophytes must settle at a high density (within 1 mm of each other) (Reed, 1990; cited in Birkett *et al*., 1998b).

Mechanised Harvesting

In Brittany, recovery appears to be sufficient to sustainably allow mechanised, commercial harvesting to re-occur every year, although concerns about declining landings have prompted some harvesters to leave *L. digitata* beds to recover for 1-2 years before re-harvesting (Werner and Kraan 2004). See below for more detail.

A study comparing hand harvesting of *L. digitata* at low tide with mechanised 'scoubidou' extraction in Brittany found no difference in recovery times after harvest (Perez 1969 In Werner and Kraan 2004).

Laminaria digitata: Ecological importance of species

Habitat provision- hodfasts

Kelp holdfasts provide refuge to a wide variety of animals supporting a diverse fauna that represents a sample of the surrounding mobile fauna and crevice dwelling organisms, e.g. polychaetes, small crabs, gastropods, bivalves, and amphipods. Kelp fronds may be colonized by encrusting bryozoans and hydroids and are grazed by molluscs such as the blue-rayed limpet *Patella pellucida*.

Habitat provision-epiphytes

L. digitata also occurs further into the intertidal than *L. hyperborea,* generally has a smooth, rather than rough stipe (which is less likely to supports epiphyte growth), and have slightly smaller holdfasts, which all drive differences in the structure and richness of associated assemblages (Blight & Thompson 2008; Teagle *et al.,* 2017). Older rougher stipes of *L. digitata* provide a substratum for a large number of epiphytic flora and fauna and it has been estimated that rugose stipes provide one and a half times that surface area provided by the bedrock (Jones *et al.*, 2000).

Habitat modification (including shading and moisture retention)

L. digitata plants are shorter, with more flexible stipes those of *L. hyperborea*. As a consequence, the fronds of *L. digitata* plants sweep the underlying rock, creating greater physical abrasion, so fewer species are able to survive in the understorey by comparison with *L. hyperborea* (Kain, 1979). This sweeping action by *L. digitata* favours a distinct species assemblage, with the limpet *Patella ulyssiponensis* and the *sponge Halichondria panacea* able to thrive, when in association with *L. hyperborea* these species are outcompeted by understory red algae.

Food provision

Species able to graze directly on the kelp include the gastropods: *Gibbula spp., Littorina spp., Patella pellucida, Lacuna spp. and* the *Rissoidae,* together with some amphipods and isopods. *Patella pellucida* forms pits similar to the home scars of intertidal limpets. The larger, laevis form excavates large cavities in the holdfast of *Laminaria* spp. which creates tissue damage weakening the adult plant and possibly contributes to its loss due to wave action and storms (Kain, 1979).

Primary production and trophic transfer

Kelp plants are the major primary producers in the marine coastal habitat. Within the euphotic zone kelps produce nearly 75% of the net carbon fixed and large kelps often produce annually well in excess of a kilogram of carbon per square metre of shore. However, only about 10% of this productivity is directly grazed. Kelps contribute 2-3 times their standing biomass each year as particulate detritus and dissolved organic matter that provides the energy supply for filter feeders and detritivores in and around the kelp bed. Dissolved organic carbon, algal fragments and microbial film organisms are continually removed by the sea. This may enter the food chain of local subtidal ecosystems, or be exported further offshore. Rocky shores make a contribution to the food of many marine species through the production of planktonic larvae and propagules which contribute to pelagic food chains.

Laminaria digitata: Ecological implications of harvesting

No evidence was found on the impacts of hand harvesting in Wales and England.

Habitat modification (including shading and moisture retention)

Canopy removal of *L. digitata* has been shown to reduce shading, resulting in the bleaching of sub canopy algae (Hawkins & Harkins, 1985). Harvesting may also result in habitat fragmentation, a major threat to this biotope’s ecosystem functioning (Valero *et al.*, 2011).

Changes in community composition

Following clearance of small experimental plots, Engelen *et al.* (2011) observed changes in the species composition (a shift towards *Saccorhiza polyschides*) and size structure of *L. digitata* for 18-24 months following clearance.

In Nova Scotia following experimental canopy clearance, Smith (1985) found only weak changes in the understory community composition, with increased growth of understory algal species in cleared plots within the first year, however this was attributed to a variety of factors in addition to clearance.

In Brittany, where *L. digitata* is at its most southerly range, it is increasingly being outcompeted by the fast growing kelp *S. polyschides* after mechanised harvesting (Werner and Kraan 2004; Mac Monagail *et al.,* 2017). This is consistent with research from clearance plots in the same region (Engelen *et al.,* 2011). If hand harvesting causes the *L. digitata* canopy to become more fragmented and disturbed, it could lead to an increased abundance of *S. polyschides* (as has been observed in France) or facilitate the spread of *U.pinnatifida* and/or L. *ochroleuca* further into Wales and England.

Commercially harvested species

Debate exists on whether kelp harvesting is detrimental to fish stocks. While some state that no negative consequences of harvesting have been documented (Vea & Ask, 2011); others suggest that as important foraging and nursery grounds for fish (and birds), removal will inevitably result in negative consequences (Lorentsen *et al.*, 2010). The impacts of removal of beach-cast weed are discussed in the main body of the report.

Relevant studies or monitoring

In Brittany, France, *L. digitata* is mechanically harvested using a 'scoubidou trawl', a sickle-like implement attached to pole that rotates around the fronds to uproot them and pull kelp aboard the havresting vessel (Mac Monagail *et al.,* 2017). The harvesting boats operate in depths of 3-5 meters, and each has a loading capacity of 10 – 20 tonnes (Werner and Kraan 2004). Each lift of the scoubidou can extract ~ 10kg kelp. The method is fairly size-selective and young kelp smaller than 60cm (<2 years old) are generally left intact, which facilitates regrowth of the canopy (Werner and Kraan 2004). Using this method, over 30% of *L. digitata* standing stock in Britany can be harvested in a single year, which combined with natural losses (e.g. due to storms), can remove up to 50% of the population (Arzel 1998, In Werner and Kraan 2004). The standing stock has remained fairly stable for decades, but landings have declined as a result of changes in the number and types of vessel operating.

Arzel, (1998), Birkett *et al.* (1998), Kain, (1975; 1979), and Markham & Munda, (1980) all studies recovery of cleared plots (see Recovery section above).

Species specific management approaches

- Take only the upper part of the frond, leaving some tissue at the base of the blade (the meristem)
- Havest during spring to avoid the reproductive season
- Harvest areas sparsely to preserve ecosystem services

NUI Galway recommend leaving at least 20cm of blade above the stipe and avoiding harvest of fertile material (Edwards *et al.,* 2012).

Biomass stock assessments have not been conducted for this species in Wales and England. Commercial harvesters monitor harvested volumes and effort; however, these figures are rarely published.

Access to this seaweed resource from the shore is limited by the tidal window, which could act as a buffer against excessive harvesting. However, should use of snorkel or SCUBA increase, harvesting pressure would increase correspondingly, potentially requiring additional management.

A number of management measures are in place for harvesting *L. digitata* with the scoubidou trawl in Brittany, including harvesting quotas (bound to each vessel), and restrictions on the number of days a week a vessel can operate in the harvesting season, which runs from the middle of May to mid-October (Werner and Kraan 2004). Less harvesting is permitted at the start of the season (2 days per week per boat) compared to the end of the season (5 days per week per boat), to limit the impact on *L. digitata* during its growing and reproductive seasons. In addition, harvesting is severely restricted in winter and only a few boats still operate to ensure supply of seaweed to domestic alginate industry (Werner and Kraan 2004). Each harvesting vessel must obtain a licence from the government, which is linked to the boat and the skipper, is non-transferable, and must be renewed annually. The coast of Brittany is divided into harvesting areas for *L. digitata,* and fishermen are usually allocated to specific areas to balance available resources and fishing effort (Werner and Kraan 2004). The state of the stock is routinely monitored by government agencies (several times per year at various location).

There are no statutory ‘fallow periods’ in place for harvesting *L. digitata* in Brittany due to its fairly rapid regrowth, so beds are typically re-harvested each year. However, concerns about declining landings resulted in some local harvesters self-managing their fishery, and leaving *L. digitata* beds to recover for 1-2 years before re-harvesting (Werner and Kraan 2004), it has been advised that leaving stocks to recover for 3 or more years after harvest would be beneficial to the long-term productivity of the fishery (Werner and Kraan 2004).

Laminaria digitata: Knowledge Gaps

- Baseline data on standing stock biomass
- Monitoring data to inform sustainable harvest practices

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Appendix 8 *Laminaria hyperborea*

Names

Laminaria hyperborea

Kombu, Forest kelp, Curvie, Redware, Sea rod, Mayweed, Slat mara

Laminaria hyperborea: harvesting

UK populations of *L. hyperborea* are currently subject to small scale hand harvesting using knives and scissors but wild harvesting activities are increasing (Marine Scotland, 2016). Hand harvested kelps are used for food, fodder, fertilizer, cosmetics and pharmaceuticals.

L. hyperborea is also harvested for alginate extraction, to produce food additives, emulsifiers, and gelling agents. The large volume required for alginate production generally requires mechanised harvesting methods, with key commercial producers including Norway and Brittany, with proposals for Scotland and Ireland (for reviews see Guiry & Blunden, 1991; Wilkinson, 1995 and Birkett *et al.*, 1998b).

The species probably contributes substantially to the volume of beach-cast kelp washed up on the shore, which is gathered for use as a soil conditioner or agricultural fertiliser. Collection of drift weed is seasonal following winter storms, and is not as widely reported from Wales and England as in Ireland or Scotland, where it is traditionally spread on machair land (Angus, 2017).

Laminaria hyperborea: Natural range / distribution

L. hyperborea is found from the extreme low water mark to depth, where it grows attached to bedrock or other stable hard substrata, forming dense forests under suitable conditions. Maximum depth is determined by light penetration except in the presence of grazing e.g. by Echinus in the Isle of Man (Jones & Kain, 1967; Kain *et al.,* 1974). This is typically about 8-12 m depth in coastal waters, up to 47 m in the clear waters around St Kilda.

It grows in exposed to very exposed wave energy, fully saline conditions. Tidal strength preferences are for weak (< 1 knot or <0.5 m/sec.) to moderately strong (1 to 3 knots (0.5-1.5 m/sec.) water flow. Intolerant to influenced by sediment (e.g. sand) scour, extreme wave action or currents (e.g. surge gullies) and high irradiance (Birkett *et al.*, 1998b; Kain *et al.*, 1975).

Found throughout Britain where the habitat is suitable. Scarce along the South-east coast of Britain due to a lack of hard bedrock substrata (See Figure 8).

Also found throughout the North-east Atlantic from the northern coast of Iceland, north to the Russian coast near Murmansk and south to Cape Mondego, mid-Portugal including Norway, Faroes, northern France and northern Spain but absent from the Bay of Biscay.

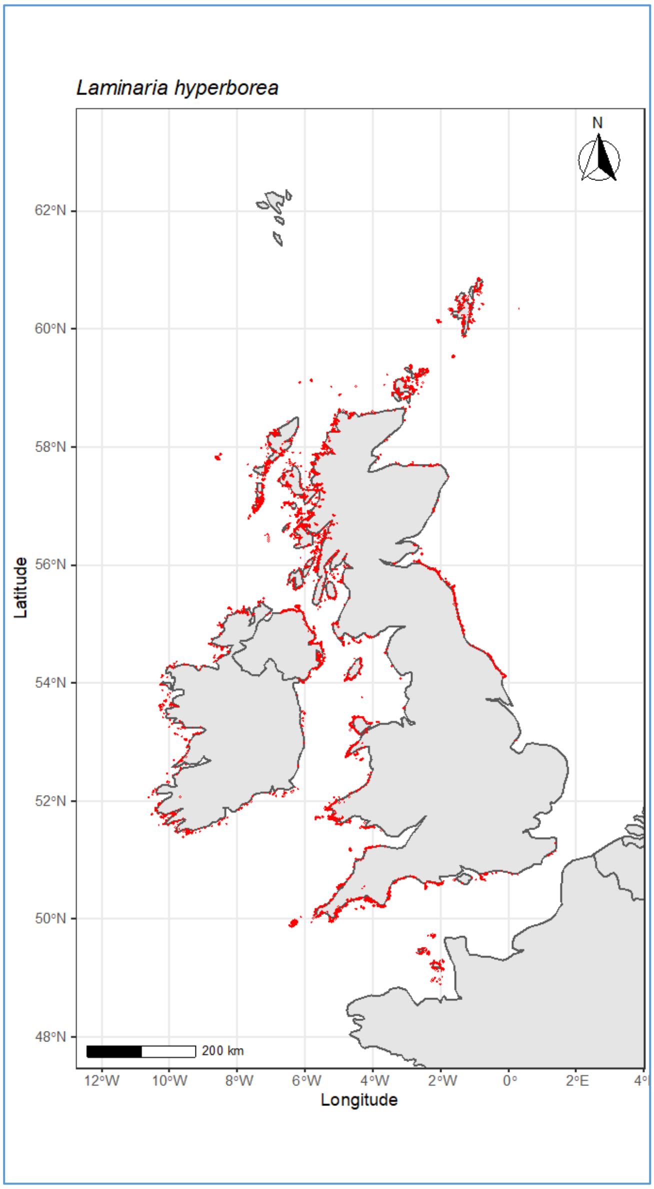

Figure 8. Distribution of *L. hyperborea* in the United Kingdom and Ireland. Occurrences of *L. hyperborea* plotted as red points. Spatial data of species occurrences obtained from NBN Atlas, DASSH and Marine Recorder databases.

Laminaria hyperborea: Growth, reproduction and seasonality

Reaches a maximum length of 3.5 m and an age of 11-20 years. Laminarian kelp all exhibit a bi-phasic lifecycle, alternating between a visible sporophyte (sometimes referred to as a “plant”), which is usually of considerable size and a microscopic, filamentous gametophyte. Juvenile *L. hyperborea* sporophytes grow rapidly for about five years, reaching maturity is 2-6 years old. It reaches greater size and age in the northern parts of its range where waters are cooler (Teagle and Smale, 2018; Sjøtun *et al.*, 1993).

Growth of the sporophyte is seasonal, peaking at a maximum rate of 0.94cm/day from November through the winter. Blade growth occurs from the base with old material shed from the tips in spring and early summer, referred to as the “May cast”, which produces large amounts of detrital material, some of which will be washed ashore as beach cast weed. Growth stops in summer around June, with carbohydrates being stored for use during the growing season (Sjøtun, 1996).

Fertile “sorus” material is formed on the blade, from which zoospores are released over a 6-7 week in winter (September - April) (Kain, 1975), from which large numbers (>1,000,000) of zoospores are released. Zoospores develop into either male or female gametophytes, in favourable environmental conditions reproduction occurs within 10 days; then the male gametes fertilise the female, and a sporophyte develops in situ. Fertile material may be found year round, peaking in late autumn to winter. Correspondingly, appearance of most juvenile sporophytes occurs in spring, although they can be found all year round (Birkett *et al.*, 1998b).

For fertilisation to be successful, gametophytes must settle at a high density (within 1 mm of each other) and therefore may suffer from dilution effects over distance (Reed, 1990; cited in Birkett *et al.*, 1998b). If conditions are not favourable, the microscopic gametophytes can survive for serval years in a state of vegetative reproduction, creating a “seedbank” from which new juvenile kelp can appear once optimal conditions return.

The chemical composition of seaweeds varies seasonally (Manns *et al.,* 2017; Schiener *et al.,* 2015), so the requirement for end use (i.e. nutrition, flavour, concentration of bioactives or carbohydrates) balanced with the need for maximum yield will influence harvest time.

Recruitment and recovery potential

No evidence was found specific to hand harvesting practices which only remove a portion of the blade. However based on the available evidence it is likely that, provided the blade is cut at least 10cm above the meristem, recovery will be rapid (within one year, or less during the winter growing season).

Kitching (1941) found that, following cutting of stipes with shears, *L. hyperborea* plants dies and were replaced within one year with a canopy of small plants. This harvest induced change in age structure and plant size is likely to have effects on the supported ecosystem.

If environmental conditions are favourable *L. hyperborea* can recover following disturbance events reaching comparable plant densities and size to pristine beds within 2-6 years (Kain, 1979; Birkett *et al.*, 1998b; Christie *et al.*, 1998). Holdfast communities may recover in 6 years (Birkett *et al.*, 1998b). Full epiphytic community and stipe habitat complexity regeneration require over 6 years (possibly 10 years). These recovery rates were based on discrete kelp harvesting events. Recurrent disturbance occurring frequently within 2-6 years of the initial disturbance is likely to lengthen recovery time (Birkett *et al.*, 1998b, Burrows *et al.*, 2014). Kain (1975a) cleared sublittoral blocks of *L. hyperborea* at different times of the year for several years. The first colonizers and succession community differed between blocks and at what time of year the blocks were cleared. *L. hyperborea* returned to near control levels of biomass within 3 years at 0.8 m but that recovery was slower at 4.4m. However, grazing slows recovery, as while spores still settled few juvenile kelps survived after 1 year in the presence of sea urchin *Echinus esculentus* (Kain, 1979). These experiments did not remove the microscopic gametophyte 'seed' bank.

Research on harvested populations of *L. hyperborea* in Norway suggests that kelp forest biomass returned to pre-harvesting levels after 1-2 years, but that the plants were mainly small (1m) and that the age structure of the population was shifted towards younger plants. Sivertsen (1991, cited in Birkett *et al.*, 1998b) showed that kelp populations stabilize after about 4-5 year post-harvesting. Re-growth was due primarily to growth of viable juveniles after harvesting. Therefore, recovery is dependent on the depth (light availability) and grazing.

Christie *et al.* (1998) found that in south Norwegian *L. hyperborea* beds a pool of small (<25 cm) understorey *L. hyperborea* plants persist beneath the kelp canopy for several years. The understorey *L. hyperborea* sporophytes had fully re-established the canopy at a height of 1m within 2-6 years after kelp harvesting. Within 1 year following harvesting, and each successive year thereafter, a pool of *L. hyperborea* recruits had re-established within the understorey beneath the kelp canopy. Christie *et al.* (1998) suggested that *L. hyperborea* bed re-establishment from understorey recruits (see above) inhibits the colonization of other kelps species and furthers the dominance of *L. hyperborea* within suitable habitats, stating that *L. hyperborea* habitats are relatively resilient to disturbance events.

Regional variation

However, recovery rates from Norway may not be representative of Wales and England, as populations of *L. hyperborea* have been found to attain greater size and presumably growth rates in higher latitudes (Teagle and Smale, 2018). Similarly, environmental conditions are likely to have a large impact of recovery potential, kelp recruitment and settlement is affected by wave exposure (Graham *et al.,*1997) and for example, the average stipe length of mature *L. hyperborea* plants can more than double along a wave exposure gradient (Smale *et al.*, 2016).

Dispersal potential

Spores may be transported at least 5 km from the parent (Jónsson, 1972, cited in Norton, 1992). They settle on the available rocky substrata after 24 hrs (Kain, 1964), although this may be longer depending on local currents (Fredriksen *et al.*, 1995). For fertilisation to be successful, gametophytes must settle at a high density (within 1 mm of each other) and therefore may suffer from dilution effects over distance (Reed, 1990; cited in Birkett *et al.*, 1998b).

Given the potentially large number of spores and gametophytes it is likely that recolonization would occur rapidly and sporophytes may grow up to 0.94 cm /day under optimal conditions, provided that adult sporophyte populations are present nearby. However, repeat harvesting may reduce the resilience of the population. Further, while the kelp biomass may recover within a number of years, there is evidence that the associated community does not (see below).

Recovery of associated species

Holdfast communities may recover in 6 years (Birkett *et al.*, 1998b), Detailed studies in Norway by Rinde *et al.* (1992, cited in Birkett *et al.*, 1998b) examined recovery of non-kelp species. The epiphyte community in control areas about 10 years old was richer and more extensive than on replacement plants in harvested areas. Of the epifauna, Halichondria sp. were only found on 10 year old plants and tunicates on plants 6 years post harvesting. Holdfast fauna was more abundant richer in 10 year old plants in control areas than younger plants in previously harvested area. Overall his results suggest that full biological recovery, or maturation, may take at least 10 years.

Laminaria hyperborea: Ecological importance of species

Relative to other seaweed species in Wales and Wales and England, there is extensive evidence of the ecological importance of this species. *L. hyperborea* is arguably to most important seaweed species for provision of ecosystem services in the UK. For the purposes of this dossier, habitat provision have been detailed below.

Habitat modification (including shading and moisture retention)

In a recent review, Teagle *et al.* (2018) outlined that kelp canopies alter light (Connell, 2003a), sedimentation (Connell, 2003b), physical abrasion (Irving and Connell, 2006), flow dynamics (Eckman *et al.*, 1989), substratum availability and condition (Christie *et al.*, 2007) and food quantity and quality (Krumhansl and Scheibling, 2012).

Habitat provision

Arguably the most crucial function performed by *L. hyperborea* is habitat provision, recently reviewed by Teagle *et al. (*2017). This ecosystem service is delivered at a level well above that of other kelp species and is supported by a considerable body of evidence (King *et al.,* 2021; Schultze *et al.,* 1990; *Steneck et al., 2002;* Teagle *et al.,* 2018; Christie *et al.,* 1998), summarised below.

The kelp forms dense stands, referred to as forests, which are diverse, species rich habitats supporting over 1,800 species in the UK (Birkett *et al.*, 1998b). These communities are generally dominated by mobile invertebrates including copepods, polychaetes, gastropods and amphipods, and by sessile fauna such as bryozoans, bivalves and sponges (Teagle *et al.,* 2017: Anderson *et al.*, 2005; Arroyo *et al.*, 2004; Blight and Thompson, 2008; Christie *et al.*, 2003; Christie *et al.*, 2009; Moore, 1972a; Norderhaug *et al.*, 2002; Ojeda and Santelices, 1984; Rios *et al.*, 2007; Schaal *et al.*, 2012). At an individual level, research from Norway has demonstrated that a single *L. hyperborea* plant supports an average of 130 species from 8000 individuals (Christie *et al.*, 2003).

L. hyperborea associated communities are variable across both local (eg driven by wave exposure, habitat structure or sedimentation) and regional (eg driven by climate) scales (Teagle *et al.,* 2018; Christie *et al.,* 1998; 2003; Smale *et al.*, 2016). This may be driven in part by morphological responses to conditions in the kelps themselves affecting the volume or structure of the available habitat (Norderhaug *et al.*, 2007; Smale *et al.*, 2016). There is some evidence that the species richness and biodiversity value of *L. hyperborea* assemblages increases with decreasing latitude (i.e. form cooler, northern waters to warmer, southern waters) (Smale *et al.,* 2020; Bue *et al.,* 2020).

L. hyperborea provides three separate micro-habitats: the blade (or frond), the stipe and the holdfast as well as understorey flora and fauna.

Habitat provision- Understorey flora and fauna

Mobile invertebrate species supported by *L. hyperborea* and prey species for fish and crustaceans, some of which are of commercial or conservation importance (Norderhaug *et al.*, 2005; Smale *et al.*, 2013; Steneck *et al.*, 2002), with a positive relationship between the abundance of fisheries resources and the extent of kelp forest (Bertocci *et al.*, 2015)

Habitat provision- holdfasts

Kelp holdfasts are complex, stable, 3-dimensional “root-like” structures used to anchor the plant to the seabed. Holdfast size and morphology are influenced by environmental conditions and are highly variable, which is reflected in the diversity and abundance of associated assemblages (Christie *et al.*, 2003; Norderhaug *et al.*, 2007; Moore, 1978).

Holdfasts have been the focus of the majority of studies and generally support higher levels of diversity (but at lower abundances) than the stipe or blade (Teagle *et al.,* 2018; King *et al.,* 2021), with reported species richness typically reaching 30–70 species per holdfast, but potentially up to 90 species (Christie *et al.*, 2003; Jones, 1972; Moore, 1972a; Thiel and Vásquez, 2000). The abundance of associated invertebrates can be more than 10,000 individuals per holdfast (Christie *et al.*, 2003; Schaal *et al.*, 2012).

Habitat provision- epiphytes

Stipe

In addition to creating habitat as a primary foundation species, *L. hyperborea* has a rough, textured stipe which supports dense stands of epiphytes, often characterised by red seaweeds, representing secondary foundation species that provides further habitat complexity, food and shelter for an abundance of associated invertebrates, increasing local diversity and richness (Christie *et al.*, 2003; Thomsen *et al.*, 2018). The species richness of these epiphytic red algae communities is highly variable, with 40 red algae species found in Norway (Christie *et al.*, 1998; Sørlie, 1994), but only four species accounting for 95% of epiphyte biomass found is samples from southeast Scotland (Whittick (1983)

King *et al.* (2021) recorded highly diverse and abundant stipe associated communities from British *L. hyperborea,* finding 134 species (87 mobile and 47 sessile) composed of 26300 individual mobile invertebrates and 15 kg of sessile organisms from 60 stipes collected from 12 British sites, although there was also high variability between individual stipes and between sites from the same region. Based on the density *L. hyperborea*, this equates to an estimated abundance of 13147 individuals per m^2 for mobile invertebrates and a biomass of 172 g m^{-1} for sessile species. These values are thought to be a conservative underestimate (King *et al.,* 2021). In Norway, Christie *et al.* (2003) found a total of 199 taxa associated with *L. hyperborea* stipes.

Assemblages associated with English (South Devon) *L. hyperborea* supported proportionally more polychaetes than Welsh (Pembrokshire) and Scottish sites, which were more dominated by amphipods (King *et al.,* 2021). Stipe-associated invertebrates represent an important resource for mobile predators and serve as a direct link to higher trophic levels (Norderhaug *et al.*, 2005). For example, fish abundances are positively correlated with stipe density, in some systems (Holbrook *et al.*, 1990; Davenport and Anderson, 2007; Bertocci *et al.*, 2015).

Blade

Recent work on four kelp species by Arnold *et al.* (2016) reported a maximum of just five or six sessile invertebrate species attached to kelp blades, which were predominantly bryozoans. *Membranipora membranacea* has been noted to be one of the few, often the only, species of sessile fauna associated with the blade of *Laminaria* species (Seed and Harris, 1980).

The blue-rayed limpet, *Patella pellucida,* is a common and locally abundant grazer found on *Laminaria* spp., where it feeds predominantly on the kelp tissue (Christie *et al*., 2003; Vahl, 1971)

A study of *L. hyperborea* along an extensive stretch of the Norwegian coastline found no species associated solely with the blade, but that around 70 species were exclusively associated with either the holdfast or the epiphytes on the stipe (Christie *et al.*, 2003). This pattern has also been shown in other studies of *L. hyperborea* (Norton *et al.*, 1977; Schultze *et al.*, 1990). It is important to note that these patterns are consistent in highly mobile groups that have the means to move throughout the entire plant (Christie *et al.*, 2003).

Laminaria hyperborea: Ecological implications of harvesting

Impacts of harvesting will be expected for the duration of recovery time. Wild (mechanised) harvesting can reduce the biomass of epiphytes associated with *L. hyperborea* by 87% (Norderhaug *et al.*, 2020) and current management approaches do not facilitate full recovery of these communities following harvesting (Christie *et al.*, 1998; Steen *et al.*, 2016).

Relevant studies or monitoring

Extensive evidence exists on the ecological importance of this species, and the impacts of mechanised harvesting from Norway. See other sections of the dossier.

Species specific management approaches

Guidance in Wales and England is summarised as follows:

- Harvest only the upper part of the frond, by cutting at a height above the meristem. Advice in Ireland specifies 20cm from the top of the stipe (Edwards *et al.,* 2012)
- Harvest during the peak growth season in the first half of the year
- Avoid harvesting during the reproductive season

- Harvest areas sparsely

Access to this seaweed resource from the shore is limited by the tidal window, which could act as a buffer against excessive harvesting. However, should use of snorkel or SCUBA increase, harvesting pressure would increase correspondingly, potentially requiring additional management.

The critical role of *L. hyperborea* in supporting highly diverse and abundant associated assemblages (Bue *et al*., 2020, King *et al*., 2021; Teagle *et al*., 2018; Smale, 2020) along the UK coastline, should be fully considered in decision-making processes for future management of these ecosystems.

Current advice in Norway suggest that kelp forest should be left for 7-10 years after harvesting for the kelp biomass and non-kelp species to recover following mechanical harvest (Birkett *et al*., 1998b).

Biomass stock assessment

Some biomass estimates exist for this species in England. Across the UK coastline, the density of mature *L. hyperborea* plants is typically ~10 plants m^{-2} (Smale *et al*., 2016; Smale and Moore, 2017). This can be used to estimate the standing stock of a given area of known extant. The estimated spatial extent of *L. hyperborea* in the NE Atlantic is ~18,000 km^{2} (Pessarrodona *et al*., 2018).

Laminaria hyperborea: Knowledge gaps

- Sustainable harvested volume for Wales and England
- Ecological impacts of hand harvest

The ecological importance of this species, and the impacts of mechanised harvesting (particularly in in Norway), are well supported by the literature. The focus of research into impacts of harvest has generally been on recovery of the resource, and not the associated community.

Limited evidence was found on the impacts of hand harvesting, which are likely to differ from mechanised methods. Caution should be exercised in generalising from other regions, as for example, kelp exhibit faster growth rates and attain greater biomass in more northerly latitudes, so are likely to recover more rapidly than in more southerly areas.

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Appendix 9 *Mastocarpus stellatus*

Names

Mastocarpus stellatus

False Irish moss, Grape pip weed, Carragheen, Irish: Clúimhín Cait (cats' puff)

Mastocarpus stellatus: harvesting

M. stellatus is harvested by plucking or cutting using small knives or scissors from the shore (Mac Monagail and Morrison 2020). In Scotland and Ireland it is also hand raked although this method is not known from England or Wales. *M. stellatus* and *C. crispus* are both harvested commercially to a larger extent in Scotland and Ireland, usually in late summer, where the stipe is removed but the base is left intact to allow re-growth.

Together with *C. crispus, M. stellatus* is harvested commercially for extraction of the phycocolloid, carrageenan, for pharmaceutical, cosmetic and food industries (reviewed by Guiry & Blunden, 1991). Carrageenan is used as an emulsifier, stabilizer, gelling or thickening agent in production of soups, jellies, ice cream, body lotions, toothpaste, beer clarification and air fresheners. It is used for curing leather and in shoe polishes. It is reported to have antimicrobial, antitumor and antiviral properties. Artisanal uses include boiling with milk and sugar or honey to serve as a drink, and as a remedy for respiratory issues are diarrhoea (Morrissey *et al.,* 2001).

While it’s primary application is for carrageenan extraction, it also has applications as animal fodder, veterinary products (to treat "wasting disease" in calves, gastric ulcers in guinea pigs, as a pregnancy dietary supplement for cows and pigs), and fertilizer (MacFarlane, 1952).

M. stellatus extracts have also been identified as a source of antioxidant, anticoagulant (Gómez-Ordóñez *et al.,* 2014) and antimalarial (Vonthron-Sénécheau *et al.,* 2011) compounds, of antimicrobial agents for fish aquaculture (Dubber and Harder, 2008), and for remediation of cadmium pollution (Herrero *et al.,* 2008)

Mastocarpus stellatus: Natural range / distribution

M. stellatus is found attached to bedrock on the mid to lower shore, where it may co-exist with *C. crispus*. It is particularly abundant in very exposed areas where it grows amongst barnacles and mussels. On less exposed shores it is often abundant under fucoids such as *Fucus serratus*. It thrives in rockpools, and can be found in the shallow sublittoral and occasionally deeper water.

No information was found within the scope of this review on the upper temperature range of this species, although it’s distribution on the shore (level with and higher on the shore) suggests that it has a higher tolerance to increased temperatures and desiccation than *C. crispus*, which can tolerate up to 20°C (Simpson & Shacklock, 1979) (although local populations may be acclimated to the prevailing thermal regime). *M. stellatus* appears to tolerate freezing temperatures (Dudgeon *et al.,* 1989, 1995).

In the Western Atlantic, *M. stellatus* can be out competed by more dominant *C. crispus*, but the two species frequently co-occur in British biotopes, although the distribution of *M. stellatus* extends further up the shore (Dudgeon *et al.,* 1989).

Found all around the British Isles but is abundant mainly on western coasts, and absent from parts of East England (See Figure 9). Also recorded from the Faeroes, Iceland, Canada and the USA.

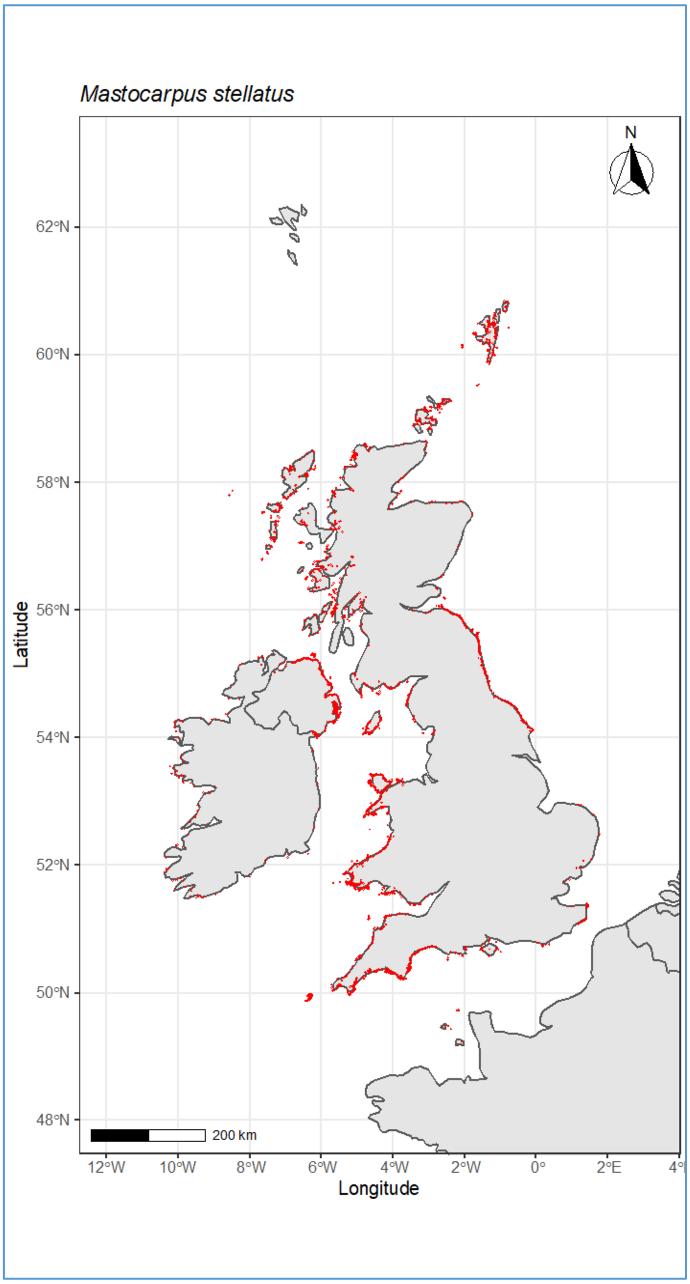

Figure 9.1Distribution of *M. stellatus* in the United Kingdom and Ireland. Occurrences of *M. stellatus* plotted as red points. Spatial data of species occurrences obtained from NBN Atlas, DASSH and Marine Recorder databases.

Mastocarpus stellatus: Growth, reproduction and seasonality

M. stellatus is a small, perennial red algae reaching up to 17 cm in length. Mature plants have conspicuous growths of short, shout papillae (reproductive bodies) on the fronds.

Measurements of in-situ growth rate in Maine, North America by Dudgeon *et al.* (1995) found a mean growth rate of 1.04 g/dry weight/m^2/day). No measureable growth of *M. stellatus* crusts was observed after a year (Dudgeon *et al.,* 1995). Growth patterns vary seasonally with the highest biomass usually in late Spring or Summer and lowest in Winter (Fernández, & Menéndez, 1991).

The life cycle of red seaweeds is complex, involving three distinct morphological stages. Alternation occurs between asexual spore producing stages (tetrasporophytes) and male and female plants producing sexually. In *M. stellatus*, the tetrasporophyte phase appears as a flat, dark red - black crust, with a rubbery appearance. This is known as the Petrocelis and capable of growing laterally and covering extensive areas. The basal crusts and crustose tetrasporphytes are perennial, tough, resistant stages that may prevent other species from occupying the rock surface and allow rapid regeneration. Although the physiological tolerances of the crust and gametophytes of *M. stellatus* vary widely (Dudgeon *et al.,* 1995) they are likely to provide a significant recovery mechanism.

The plants seen on the shore are gamtophyte stages, with separate male and female plants, which are distinguishable from one another only by microscopy. The tetrasporophyte produces microscopic tetraspores, which settle and develop into either male or female gametophytes. The female gametes are fertilised *in situ* inside the female fruiting bodies (called carposporangia) where it develops into the carposporophyte, visible with the naked eye. This cystocarp grows attached to the female plant, until it releases carpospores, which will develop into a new tetrasporophyte.

Mastocarpus stellatus: Recruitment and recovery potential

Harvesting following appropriate guidance is likely to remove only the upper part of the frond, leaving the base and holdfast from which new fronds can regenerate. New plants can also grow from the tough basal crusts, which are perennial, and may prevent other species from occupying the rock surface and allow rapid regeneration. They may therefore provide a significant recovery mechanism.

Where holdfasts and basal crusts are removed, recovery will depend on recolonization via spores. The spores of red algae are entirely reliant on the hydrographic regime for dispersal (Norton, 1992). Recruitment of *M. stellatus* is likely to occur on a very local scale, such as within 10 m of the parent plant. Recovery of a population of *M. stellatus* is therefore likely to be largely dependent on whether holdfasts remain, from which new fronds can regenerate (Holt *et al.,* 1995).

Most of the evidence for recovery of *C. crispus* is based on experiments that simulate the effects of different harvesting mechanisms and intensities (Macfarlane, 1952; Mathieson &

Burns, 1975). Due to similarities in biology, it is suggested that these recovery rates may also be applicable to *M. stellatus*:

Macfarlane (1952) in a series of experiments identified that where *C. crispus* was removed by cutting of fronds or thorough raking (leaving the crusts undamaged) the turf had recovered and there were no notable differences between the experimental areas and control sites. However, where the crusts were removed by scraping or damaged the experimental plots were still recovering nearly two years after the treatment. Following experimental harvesting by drag raking (where holdfasts and small blades were undamaged) in New Hampshire, USA, populations recovered to 1/3 of their original biomass after 6 months and totally recovered after 12 months (Mathieson & Burns, 1975). The authors suggested that control levels of biomass and reproductive capacity are probably re-established after 18 months of regrowth (where crusts are not removed). It was noted however, that time to recovery was much extended if harvesting occurred in the winter, rather than the spring or summer (Mathieson & Burns, 1975).

Minchinton *et al.* (1997) documented the recovery of *C. crispus* after a rocky shore in Nova Scotia, Canada, was totally denuded by an ice scouring event. Initial recolonization was dominated by diatoms and ephemeral macroalgae, followed by fucoids and then perennial red seaweeds. After 2 years, *C. crispus* had re-established approximately 50% cover on the lower shore and after 5 years it was the dominant macroalga at this height, with approximately 100% cover.

Similarly, MacFarlane (1952) reports that following destruction *C.* beds during a very cold winter, the annual brown alga *Chordaria* had colonized the area, followed by *Fucus. C.* did not noticeably start to grow back in the area until four years later. While *M. stellatus* is more tolerant to freezing temperatures, its recovery response following total removal may still be similar.

Pringle and Semple (1980) estimated it would take about four years for a bare patch in a *C.* bed to fill in with harvestable plants and five to ten years for *C.* to re-establish in barren areas.

Turf forming biotopes and fucoid dominated biotopes may represent alternate stable states that continue while the dominant turf or fucoids occupy space. Removal of the turf may therefore allow re-establishment of a Fucoid or kelp dominated biotope that will remain until environmental or other factors again alter the state. Lubchenco (1980) for example, on shores in New England, found that the removal of *C. crispus* turf allowed the establishment of Fucus spp. MacFarlane (1952) also recorded a shift to a *Corallina officinalis* and encrusting coralline biotope following over raking (for harvesting) of *C. crispus* turf, in these areas gastropods had increased in abundance and prevented the recovery of *C. crispus* by grazing. It should therefore be noted that where red algal turfs are removed, recovery may be prolonged.

Increased harvesting intensity has been reported to impact the size-class structure of the population by reducing mean frond length (McLaughlin *et al.,* 2006).

Based on the evidence drawn from harvesting of *C. crispus, M. stellatus* is likely to recover quickly from a single harvesting event, in situations where some holdfasts remain for regeneration of fronds. In situations where the entire population is removed, recovery will be limited by recruitment from a remote population. Repeated harvesting in the short term may impair reproductive capacity, while long term, heavy exploitation has the potential can severely deplete the seaweed resource, alter community structure and change the dominant species.

Mastocarpus stellatus: Ecological importance of species

M. stellatus and *C. crispus* are characterising species in various British biotopes. Although both species are widespread on the lower shore and sublittoral fringe, they only infrequently occur in a distinct band, or in large enough patches, to be considered as a distinct biotope "*M. stellatus* and *C. crispus* on very exposed to moderately exposed lower eulittoral rock". Both species contribute to several other biotopes as a mixed red algal turf or as co-dominat species, and *M. stellatus* can feature in high abundance.

Habitat provision

The species associated with the turf of *M. stellatus* and *C. crispus* occur in a number of biotopes and are not key characterizing species but do contribute overall to species richness, diversity and ecosystem function.

109 species of invertebrate have been recorded in association with *M. stellatus* (Schaal *et al.,* 2016), in comparison to only 34 on *P. palmata.* The authors attributed these differences to the increased structural complexity of *M. stellatus* which trapped more sediment and detritus around the base of the algae, offering greater shelter and food for small crustaceans.

Habitat modification

At high tide, dense stands of *C. crispus* mitigate water flow velocity reducing hydrodynamic stress for mobile invertebrates sheltering within the algal beds (Johnson 2001, Boller and Carrington 2006). Due to its similar morphology *M. stellatus* may perform a similar function.

Mastocarpus stellatus: Ecological implications of harvesting

Population impacts

Ecological effects may also include sub-lethal population level impacts, such as a reduction in reproductive capacity – as has been reported in *C. crispus* - while in the long term, it has the potential to alter community structure and change the dominant species.

Changes in community composition

Species which shelter within *M. stellatus* beds are likely to either be removed or displaced to some extent by harvesting activity. Due to a lack of evidence for *M. stellatus,* the information is based on morphologically similar *C. crispus*.

Commercial harvesting of *C. crispus*, a morphologically similar species, has been well studied in Canada. *C. crispus* beds were heavily exploited, resulting in declines and replacement by another red seaweed, *Furcellaria lumbricalis* (Sharp *et al.,* 1993).

Similarly, Minchinton *et al.* (1997) and MacFarlane (1952) report that following destruction *C.* beds initial recolonization was dominated by fucoids and the brown algae *Chordaria* respectively, with the perennial red seaweed taking 4-5 years to re-establish dominance. Clearly these difference in macroalgal canopy forming species will impact on biodiversity and ecological function.

In Nova Scotia, high *C. crispus* cover was associated with a greater diversity of invertebrate species but fewer algal species than low *C. crispus* cover (Scrosati., 2016). Hence, the proportion of the canopy harvested is likely to influence the composition of the associated community.

Relevant studies or monitoring

No evidence was found of the ecological importance, harvesting, or standing stock biomass of this species in England and Wales.

Species specific management approaches

- Leave the holdfast and the base of the frond intact
- Harvest seasonally (McLaughlin *et al.,* 2006)

Mastocarpus stellatus: Knowledge Gaps

No evidence was found of the ecological importance, harvesting, or standing stock biomass of this species in England and Wales. Very little is known about the basal crust (tetrasporophytic stage). Recovery and management approached are inferred from the morphologically similar *C. crispus*, with which this species co-occurs. They are likely to be harvested together in England and Wales.

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Appendix 10 *Palmaria palmata*

Names

Palmaria palmata

Dulse, Dillisk, Dilsk, Red dulse, Sea parsley, Duileasg, Chreathanch, Duileasc, Söl.

Palmaria palmata: harvesting

P. palmata is harvested both commercially and for personal consumption by hand plucking or by cutting using small knives or scissors (Mac Monagail and Morrison 2020; Scottish Government 2016). It may be collected throughout the year, although the peak is from May/June to September/October (Grote, 2019; Mac Monagail and Morrison 2020; Scottish Government 2016). Perry *et al.* (2014) state that it is at its best in spring, which may correspond with a peak in recreational harvesting.

This species has a long history of harvest by coastal communities, the practice sometimes known as ‘dulsing’, particularly from Ireland, Scotland and France. *P. palmata* is rich in fibre, vitamins and minerals, and a good source of protein. It can be eaten raw or cooked as a “sea vegetable”, but is usually dried and/or flaked for use as a condiment (Guiry & Blunden, 1991). The species is also used as fodder for animals, in cosmetics and in pharmaceuticals, for example due to its antioxidant properties (Cornish and Garbary 2010).

It is also commercially harvested in Europe, Norway, Iceland, Canada, and New England. It can be cultivated as part of IMTA systems for bioremediation purposes (e.g. Corey *et al.,* 2014).

Palmaria palmata: Natural range / distribution

P. palmata is found growing on bedrock, boulders, and epiphytically on other seaweeds from the mid to lower shore, in pools, and into the subtidal to depths of 20 m. In grows well in both sheltered and moderately exposed areas, although its form can vary widely with environmental conditions.

Where competition for space and light restricts the occurrence of *P. palmata* on rock the species often has an epiphytic habit on other algae, including kelps and fucoids, especially on the stipes *Laminaria hyperborea*.

The species is most abundant on moderately exposed to sheltered shores, although it has been recorded from highly exposed shores in Norway in association with *A. esculenta* stipes (Jorde, 1966). It is also tolerant of siltation (Irvine, 1983).

P. palmata is likely to be tolerant of small changes in salinity because as an intertidal species it is regularly exposed to precipitation, and it can cope with slightly reduced salinity under laboratory conditions (Robbins, 1978), however it is not recorded from estuarine environments.

P. palmata does well in low temperatures, with an optimum between 6 and 15°C, consistent with its distribution in northern temperate and arctic waters. In the laboratory, plants only became fertile if left at temperatures between 5-7°C with a short light period (Meer van der and Chen 1979), with mortality occurring within a week at 20°C and above (Morgan *et al.,* 1980). Such high temperatures, however, are unlikely in most parts of Britain and Ireland.

P. palmata is found in a range of water flow regimes from moderately strong to weak. When growing in steady tidal streams, where the frond is held perpendicular to the incident light, it can grow up to a meter long (Jorde, 1966).

Distributed throughout Britain and Ireland, but absent from significant stretches of coast in eastern England (See Figure 10). Also found through Arctic Russia to northern Portugal; from Artic Canada to USA (New Jersey) in the western Atlantic; and from USA (Alaska to California), Japan, Korea in the Pacific.

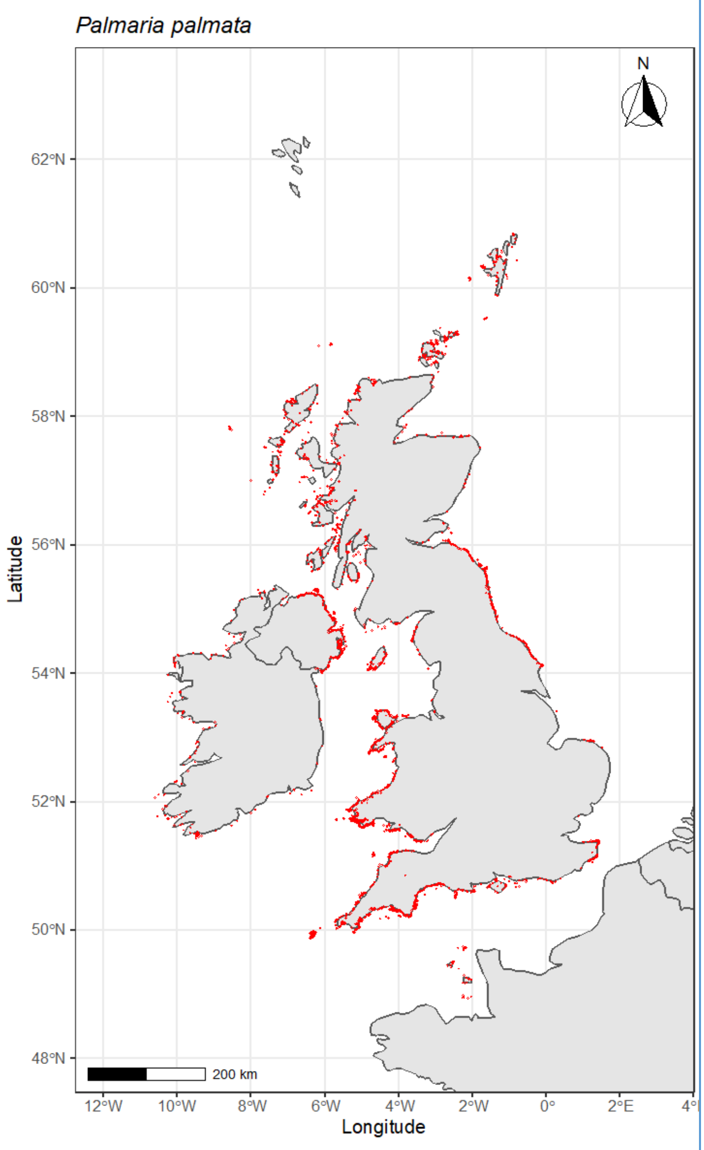

Figure 10. Distribution of *P. palmata* in the United Kingdom and Ireland. Occurrences of *P. palmata* plotted as red points. Spatial data of species occurrences obtained from NBN Atlas, DASSH and Marine Recorder databases.

Palmaria palmata: Growth, reproduction and seasonality

P. palmata is a perennial species. New frond growth occurs every year from the holdfast, which can remain for several years. The frond usually reaches between 20 and 50 cm in length, but sometimes up to 1m. Frond width is about 3-8 cm, and rarely up to 30-40 cm, reaching greater lengths in exposed conditions (Edwards *et al.,* 2012). Older fronds may grow small 'leaflets' along the margin especially where damaged, and new tissue is formed from the tips of the frond, rather than the base (Edwards *et al.,* 2012).

Growth rates can be in excess of 4% per day in Canada, although this region may experience significantly faster rates than in the north east Atlantic (e.g., Spain, N. Ireland) (Lukeman *et al.,* 2012). On the northern coast of Spain, populations of epiphytic *P. palmata* showed a seasonal growth cycle with net growth from March-August and breakage from August to March (Faes & Viejo, 2003). In Ireland, a doubling of frond length was observed every 2-4 weeks during summer (Werner and Dring, 2011). In culture, growth rate was found to vary seasonally, being highest from April to June (at 8.0 - 9.0 °C), dropping to zero above 14 °C, with disintegration of the frond observed in association with reproductive maturity through the summer (Corey *et al.,* 2014).

The life cycle of *P. palmata* is complex and unusual, consisting of reduced female gametophyte, a macroscopic male gametophyte and a foliose tetrasporophyte (an asexual spore producing stage). The male gametophyte and tetrasporophyte are indistinguishable from one another prior to maturity. Male and female plants are separate, with female gametophytes having a thin, crust-like form, and male gametophyte growing as a macroscopic foliose form. Male gametophyte plants are relatively scarce, so the majority of recognisable seaweed ‘plants’ seen on the shore are likely to be tetrasporophytes.

Spores are produced from November and December until March (Werner and Dring, 2011). Microscopic tetraspores are then released, which settle and develop into either male or female gametophytes. The male gametophyte produces spermatia that can fertilize the female *in situ*. Following fertilisation, the young blade-like tetrasporophyte grows attached to the female gametophyte, later forming its own basal system which completely overgrows the tiny female.

Age at maturity is different for male and female plants. Male became fertile within 9-12 months, while females need only a few days to become sexually mature. In the laboratory, plants only became fertile if kept at temperatures between 5-7°C with a short day light period (Meer van der and Chen 1979).

Palmaria palmata: Recruitment and recovery potential

Recovery of *P. palmata* can occur through regrowth of damaged fronds, regrowth from surviving perennial holdfast or recolonization by propagules. Recovery of this species is likely to be highly dependent on the severity of the impact (whether the holdfasts and basal crusts are removed or damaged) and its spatial footprint. The rapid colonisation by *P. palmata* observed by Hawkins & Harkin (1985) suggests that, where source populations are nearby, dulse is a relatively opportunistic species, appearing before fucoids in the process of succession.

Where only part of the frond is harvested, regrowth will be rapid during the spring and summer growth season and older parts of individuals typically have small 'leaflets' along the margin from regeneration. Due to its seasonal growth pattern (Faes & Viejo, 2003), regeneration is likely to take longer between August and March.

Following clearance of the kelp canopy, a rapid increase in the number of *P. palmata* sporelings was observed, and the species came to dominate cleared plots within five months (Hawkins & Harkin 1985). In this case it is likely that recovery was high because *P.*

palmata present epiphytically on adjacent kelp plants may have supplied recruits (Hawkins & Harkin, 1985). Also, basal crusts and microscopic life stages are likely to have been left intact by kelp clearance. The experimental results suggest that small gaps (in this study 2 x 2 m) can be filled rapidly where adults remain, however these results should not be generalised to impacts where removal occurs over a wide area.

This is consistent with observations from the north western Atlantic, that, *P. palmata* requires an intermediate level of disturbance (in this case turning over of boulders by storms) in order to reach a harvestable population density; too little and succession will proceed allowing longer lived perennials like *C. crispus* and *M. stellatus* (Garbary *et al.,* 2012) to dominate.

Dispersal distances are short (<10 m). Successful fertilisation requires male gametes to reach the female gametes *in situ* of the female gametophyte, so males release spermatia that then sink rapidly. As a result, spore dispersal is probably determined by currents and turbulent deposition (Norton 1992). Red algae produce large numbers of spores that may settle close to the adult, especially where *P. palmata* is epiphytic on kelps, the canopy of which is known to dampen water flow.

It is likely that *P. palmata* could recover quickly from a harvesting event in which only a portion of the blade is removed, leaving the holdfast. The species can also probably recolonize within a short period of time in ideal conditions, provided that adjacent populations are nearby. Since the dispersal range of spores is limited, recolonization from distant populations would probably take a long time, should overexploitation remove *P. palmata* from large patches of the shore.

Palmaria palmata: Ecological importance of species

Primary production and nutrient cycling

As a primary producer, this species is likely to play a role in production and trophic transfer, however no evidence was found on these functions within the time frame of this review.

Habitat provision

Due to the flattened nature of *P. palmata* fronds, they are likely to support relatively reduced species richness compared to many other red seaweed species (i.e. *C. crispus*), and certainly than large, long lived kelps. 109 species of invertebrate have been recorded in association with *M. stellatus* (Schaal *et al.,* 2016), in comparison to only 34 on *P. palmata.* The authors attributed these differences to the increased structural complexity of *M. stellatus* which trapped more sediment and detritus around the base of the algae, offering greater shelter and food for small crustaceans.

As the key characterizing and structuring species in the biotope "*P. palmata* on very exposed to moderately exposed lower eulittoral rock" *P. palmata* clearly plays a valuable role in delivery of habitat provision, particularly on moderately exposed shores in which a diverse macro algal assemblage is unlikely to be highly abundant. Christie *et al.* (2007) found that the gastropods *Ansates pellucida* and *Lacuna vincta* were the dominant fauna on *P. palmata* from kelp stipes in Norway, but limited evidence was found on associated species for Wales and England, beyond those recorded in the biotope.

Palmaria palmata: Ecological implications of harvesting

Garbary *et al*. (2012) studied populations harvested by hand-picking in comparison with unharvested populations in Nova Scotia. The observed a dense cover of *P. palmata* on many of the rocks on harvested shores. Non-harvest shores consisted of boulders which were of different size categories than harvested shores, and when *P. palmata* was present it was typically epiphytic on other algae. While there was little difference in average cover of *P. palmata* between harvest and non-harvest shores, they found that on harvested shores, on individual boulders where the species occurred, it covered >90% of the boulder surface, while adjacent rocks had very little cover. They also found that frond length was greater, and frond density was three times higher on harvested shores than on the non-harvested shores. These observations are consistent with those conducted more recently by the Cornish Seaweed Company, who monitor their harvests in order to ensure sustainability. For *P. palmata*, they have observed with annual harvesting that “the more you take the more it seems to grow”, although they are careful to leave unharvested areas as a source population (Cornish Seaweed Co. pers comm. Wilding 2021)

Stagnol *et al.* (2013) found that hand harvesting of *P. palmata* did not have a significant impact on the diversity of the site during a 12 month period, however in this case *P. palmata* was epiphytic on a canopy of *Fucus serratus* which was not removed by the harvesting activity (Stagnol *et al.,* 2013).

Relevant studies or monitoring

Using mathematical modelling, Lukeman *et al.* (2012) suggest that current harvesting activity in Canada is sustainable. They calculated that the fresh biomass of harvested *P. palmata* was about 50% of the total resource, which was about 1,600 g m-2. With growth rates in excess of 4% per day and a 50% harvest of the standing crop each month, the model suggests that the resource is sustainable at current harvest levels.

Garbary *et al.* (2012) also conducted experimental removal of *P. palmata* and assessed simulated removal of *P. palmata* by an experienced commercial harvester. Simulated commercial harvesting reduced cover of *P. palmata* from 70% to 40%, although experimental removal on shores not usually harvested reduced cover to 20% (Garbary *et al.,* 2012).

Species specific management approaches

- Leave the holdfast and a portion of the blade intact

In the western Atlantic, average harvest on the Nova Scotia coast of the Bay of Fundy from 1982-1991 was 2 dry tonnes. During the same time frame, New Brunswick shores of the Bay of Fundy had an average yield of 38 dry tonnes. This difference between the two sides of the Bay Fundy suggests that either the resource in Nova Scotia is much less, or it is not being harvested to its maximal potential (Chopin and Ugarte 2006).

Lukeman *et al.* (2012) state that, while policies for sustainable development, integrated management, and precautionary approaches are in place under the Canadian Oceans Act (Sharp and Bodiguel 2001 cites in Lukeman *et al.,* 2012), there are no legal requirements or management of Dulse harvesting. While there is some general regional variation, harvesting generally takes place only on the lowest spring tides (i.e. 7 days per month), the activity focussed in the summer months (Chopin and Ugarte 2006). Harvesting is by hand, and only the longer, easily picked fronds are removed, leaving most of the holdfasts and smaller fronds to regenerate. While the same beaches are harvested repeatedly over season, time for regrowth occurs between successive spring tides (Lukeman *et al.,* 2012).

Palmaria palmata: Knowledge Gaps

- Baseline standing stock biomass
- Sustainable harvesting yield / volumes for Wales and England
- Impact of increasing temperature on reproduction (Grote, 2019)

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Appendix 11 *Porphyra spp.*

Names

Reliable species differentiation of *Porphyra* spp. without molecular work is very difficult and it is most likely a species mixture is harvested.

Laver, Purple laver, Bbara lawr, Nori, Welshman's caviar, Sloke.

Porphyra spp: harvesting

There is inconsistency in the literature as to the harvesting method used: removal of entire plants (selects individuals which are larger than 5 cm) in practices in South Wales (Knoop, 2019), while scissors or a small knife may also be used to carefully cut the frond, leaving the base of the frond and the holdfast intact. commercial harvest whereas individuals are torn with holdfast commonly staying intact allowing for regrowth.

This genus has a long history of harvesting in Wales for use as Laver Bread and for personal consumption. It is also commercially harvested as a dried, flaked product, and can be made into Nori sheets for Sushi (although these are more likely to be imported from Asia than locally sourced). It is boiled and eaten as a jelly in South Wales, and reportedly eaten cold with vinegar in Cornwall. In South Africa, *Porphyra* spp. are harvested for Abalone fodder (Griffin *et al.,* 1999).

Porphyra spp: Natural range / distribution

Porphyra spp. are opportunistic, ephemeral species that are able to rapidly colonize newly created gaps across a range of sediment types, shore heights, wave exposures and current regimes. It grows attached to bedrock, stones, pebbles, mussel shells and other algae. Able to withstand prolonged periods of exposure to air, it tolerates a greater degree of wave action than most other red algae and thrives in sand scoured conditions. It occurs singly or in dense mats throughout the intertidal but most frequently at the upper levels.

P. umbilicalis is at the centre of its range in the UK, suggesting that it can withstand increases in temperature above those currently experienced in Welsh and English waters. *P. umbilicalis* species appear to be intolerant to brackish conditions: *Porphyra purpurea* experienced reduced photosynthetic rates with reductions in salinity (Reed *et al.,* 1980), and is absent from reduced salinity conditions (Connor *et al*., 2004)

Abundant on rocky shores throughout the UK and Ireland (See Figure 11). Its range extends throughout the Northern Atlantic from Iceland south to Portugal, east to Sweden, and from Canada and the USA to the west.

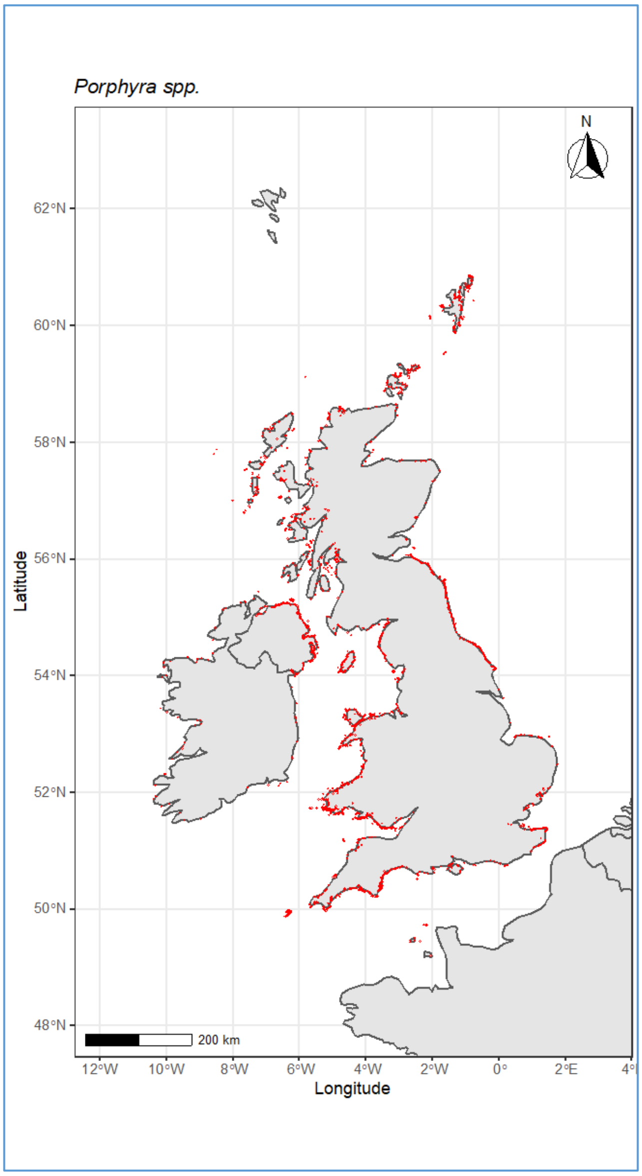

Figure 11. Distribution of *Porphyra* spp. in the United Kingdom and Ireland. Occurrences of *Porphyra* spp. plotted as red points. Spatial data of species occurrences obtained from NBN Atlas, DASSH and Marine Recorder databases.

Growth, reproduction and seasonality

Porphyra spp. may regenerate fronds from the disc shaped holdfast, and can regrow rapidly to in areas of high sand abrasion and physical disturbance. They have high growth rates and can reach a maximum size of up to 20 cm across within only a few months. *Porphyra* spp. appear to benefit from sand emersion, either through enhanced growth and/or reproduction, or by reduced competition and/or predation (Littler *et al.,* 1983; Airoldi, 2003).

In Wales growth are reproduction are highly seasonal, with high reproduction taking place during spring and recruitment during winter. Greatest cover of *P. doioica* was observed during spring and summer and much lower cover during autumn and winter (Knoop, 2019).

The reproductive output is likely to be high and the lifecycle is complex, alternating between a macroscopic gametophyte stage and a microscopic sporophyte (conchocelis stage) which burrows into wood, rock or the shells of molluscs (Anderson *et al.,* 2008). This cryptic stage enables *Porphyra* spp. to survive periods of intense disturbance or grazing which removes adult plants and allows a sudden bloom to form when conditions are suitable (Robles, 1982). Vegetative growth of the conchocelis is high during long daylight hours, while mass spore release can be triggered by a decrease in temperature (Knoop *et al.,* 2020). The harvested fronds are the gametophyte stage, for which optimal temperature, photoperiod and light intensity may depend on local conditions (Knoop *et al.*, 2020).

Porphyra spp: Recruitment and recovery potential

Following harvest fronds can regrow quickly from the disc shaped holdfast, and is able to opportunistically and rapidly recruit to cleared areas of suitable habitat. The life history characteristics that support this opportunism are the broad tolerances for a wide range of conditions (Vermaat & Sand-Jensen, 1987) and high growth and reproduction rates. This species is naturally resilient to disturbances such as sand emersion, with recovery taking place from the microscopic conchocelis stage.

Due to its seasonal life history strategy, opportunism, weak competitiveness and reliance on the availability of bare rock substrates, harvesting season is an important factor in recovery. Harvesting during the low recruitment season can potentially limit recovery and result in displacement by competing *Ulva* spp. (Knoop, 2019). In Wales, fast recovery following harvesting in spring was reported, while recovery following harvest in autumn was limited, and was accompanied by reduced yields (Knoop, 2019).

After the 1967 Torrey Canyon oil spill, off the south west coast of the UK, *Porphyra* spp. were observed within two months in areas where oil had killed the herbivores that usually grazed on them (Smith, 1968). It may regenerate from its discoid shaped holdfast if it remains in situ, or from the microscopic burrowing conchocelis stage, which can survive periods in which adult plants are removed (Robles, 1982), and may buffer populations from some level of harvesting pressure (Nelson and Conroy, 1989 IN Stagnol *et al*., 2013).

In contrast to the reported rapid recovery rates above, harvesting method was found to be an important factor for *P. umbilicalis* recovery in New Zealand (Nelson *et al*., 1989). Where the basal part of plants was left intact, re-harvest was possible within two months, while little growth was observed over the same time period following total clearance.

Recruitment and recovery may be impaired where high densities of grazers occur. Experimental manipulation of limpets and littorinid densities has shown that grazing by these species can remove significant amounts of ephemeral algae and prevent blooms forming (Lein, 1980, Robles, 1982, Albrecht, 1998, Jenkins *et al.,* 2005).

Porphyra spp: Ecological importance of species

Habitat provision

Relative to other harvested seaweeds, it is of limited habitat value due to its sheet-like, almost two-dimensional form. *Porphyra* spp. can form dense mats with green *Ulva* spp, forming a defined biotope in areas where physical disturbance from sand abrasion prevents the development of a longer-lived biological assemblage, such as fucoids, more typical of stable rocky shores. Although species are scarce and are not unique to *Porphyra* spp. habitat, the seaweed cover provides shelter and food for grazers, including the limpet *Patella vulgata* along with barnacles and occasional winkles *Littorina littorea* and *Littorina saxatilis* and many amphipods that provide shelter and protection from dessication, especially during low tide.

Porphyra spp: Ecological implications of harvesting

Seasonal variability and natural environmental variation resulting from regular sand inundation appear to be the key drivers of *Porphyra* spp., population dynamics, including percentage cover, recruitment and abundance (Knoop, 2019; Stagnol *et al*., 2013). These drivers mask any harvest induced patterns in recovery and associated biodiversity, with both authors concluding that harvesting impacts are minimal.

Habitat provision

In South Africa, harvesting was found to remove patches of *P. umbilicalis* plants. Harvesting of *P. umbilicalis* reduced populations of associated amphipods, isopods and littorinid snails, these changes were comparable to natural *P. umbilicalis* population decreases (Griffin *et al.*, 1999).

Relevant studies or monitoring

Until recently, very few studies on harvesting of this species were found form Britain, however a recent PhD thesis has addressed knowledge gaps for *P. dioica* in South Wales, towards a goal of sustainable cultivation (Knoop, 2019). Harvesting, which targets plants >5cm, was found to impact blade length resulting in a 64% reduction in blade size compared to unharvested areas. Hand harvesting was found to have minimal impact on percentage cover, yield, and associated community, while natural variation due to disturbance (sand cover) was high (Knoop, 2019).

Nelson *et al*. (1989) found that yields were greatest from harvests in the latter half of the *P. umbilicalis* growing season in New Zealand.

Harvesting of *P. umbilicalis* in South Africa reduced the standing stock biomass at a level which was detectable for up to six months, however these ecological impacts were considered comparable to natural population variation (Griffin *et al.,* 1999).

Species specific management approaches

- Leave the basal portion of the frond and holdfast intact
- Do not strip the entire plant from rocks

Based on the findings of (Knoop, 2019), harvesting is recommended to take place in spring, as opposed to autumn, to generate higher yield and faster regrowth.

Current harvesting practice in Wales involves removal of entire plants and selects for plants which are larger than 5 cm in length (Knoop, 2019). Torn individuals are left behind (Knoop, pers. comm to authors). Following the findings of Nelson *et al.* (1989), it is recommended that harvesting does not remove the basal portions of the plants.

Perry *et al.* (2014) identified concern for populations in Wales due to intensive harvesting activity, particularly at two sites: is Freshwater West and Bracelet Bay. A site specific management approach is recommended for the species at these locations.

An individual commercial forager collected between 3kg and 22.3kg of laver on any one collection day at 5 sites within Pembrokeshire Marine SAC over a 10 month period in 2013 (Perry *et al*., 2014).

Porphyra spp: Knowledge Gaps

- Standing stock biomass in Wales and England remains unknown, and is likely to fluctuate widely and seasonally
- The length of the optimal recovery period between harvests remains unknown (Knoop, 2019)

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Appendix 12 *Saccharina latissima*

Names

Saccharina latissima

Sugar kelp, Sweet kelp, Kombu royale, Sea-belt, Devils apron, “Poor Man’s Weather Glass”

Saccharina latissima: harvesting

Currently in the UK, harvesting of *S. latissima* is confined to manual harvesting on a small commercial scale. Manual harvesting involves removal of only a portion of the blade, accessed either from the shore at low tide or by boat. Diving and snorkelling practices are not widely used, however, increased hand harvesting using snorkel or SCUBA would increase harvesting pressure correspondingly, potentially requiring additional management.

Historically, this species was gathered as beach cast weed for use as a fertiliser, although the current extent of this activity in England and Wales in unknown. On the Scottish Islands of Ronaldsay and Orkney Island, there is a breed of sheep that spends several months of the year grazing on seaweed – particularly *S. latissima*.

In England and Wales today, *S. latissima* is used predominantly for human consumption due to its high nutritional value. It naturally contains high levels of minerals, sugar and monosodium glutamate, which give it a sweet, salty “Umami” flavour (Mouritsen *et al.,* 2012**).** It is usually dried and flaked, and sold as “Kombu royale”, a “sea vegetable” (Chapman *et al.,* 2015; Birket *et al.,* 1998; Rey *et al.,* 2019) or as flavour enhancer (Mouritsen *et al.,* 2012; Chapman *et al.,* 2015). Extracts of *S. latissima* are also used in the cosmetic industry.

Due to its high carbohydrate content, there has also been considerable interest in this species for production of biofuels (Kerrison *et al.,* 2015). However, for this bulk application, *S. latissima* is typically produced by cultivation methods rather than wild harvested (Handå *et al.,* 2013).

Saccharina latissima: Natural range / distribution

S. latissima usually grows on hard substrate from bedrock and large boulders down to pebbles. Even sand grains can provide an attachment substrate sufficient for juveniles to develop into new plants, and large, 'loose lying' populations have been recorded in calm conditions, with no signs of ever having been attached to hard substrate (Burrows, 1958). The species is most abundant on sheltered shores from the lower shore into the shallow sublittoral, in areas with fast moving currents. It can tolerate some wave exposure, with the flexible stipe reducing leverage on any substrate to which it is attached, but is absent from the intertidal at exposed sites, in which it may extend into the lower eulittoral where the impact of wave action is reduced (Birkett *et al.,* 1998). It may be found from weak < 1 knot (<0.5 m/sec.) tidal flows, but thrives stronger currents and rapids systems.

Growth is optimal at full salinity, although plants can survive reduced salinities, albeit with reduced growth rates (Kain, 1979). *S. latissima* is a cold-temperate species that can acclimate to a wide range of temperatures (Anderson *et al*., 2013) and local thermal adaptations have been reported, with good germination at 0°C (Sjøtun & Schoschina, 2002), optimal growth at 10-15 °C (Bolton & Lüning, 1982) and an upper temperature limit of 17-22 °C (Gerard & Du Bois, 1988; Lee & Brinkhuis 1988). Ecotypic differentiation of populations has been reported with respect to light and temperature along its latitudinal range (Heinrich *et al.,* 2012).

S. latissima is common around the coasts of the British Isles (See Figure 12). It is widely distributed throughout the north Atlantic from Svalbard to Portugal (Birket *et al.,* 1998; Connor *et al.,* 2004; Bekby & Moy 2011; Moy & Christie 2012), through the Barents Sea and Baltic Sea, but is apparently absent from the Bay of Biscay. Also from Alaska through California in the Pacific and through Canada as far south as New Jersey in the Western Atlantic.

The shape of the frond can vary with environmental conditions, becoming narrower and more streamlined when subjected to higher water flow rates (Gerard, 1987). Plants from wave exposed sites have short, solid stipes and short, narrow and thick fronds with closely wrinkled blades (Lüning, 1990). In contrast, plants from sheltered sites have a broad thin blade with an undulate surface (Lüning, 1990).

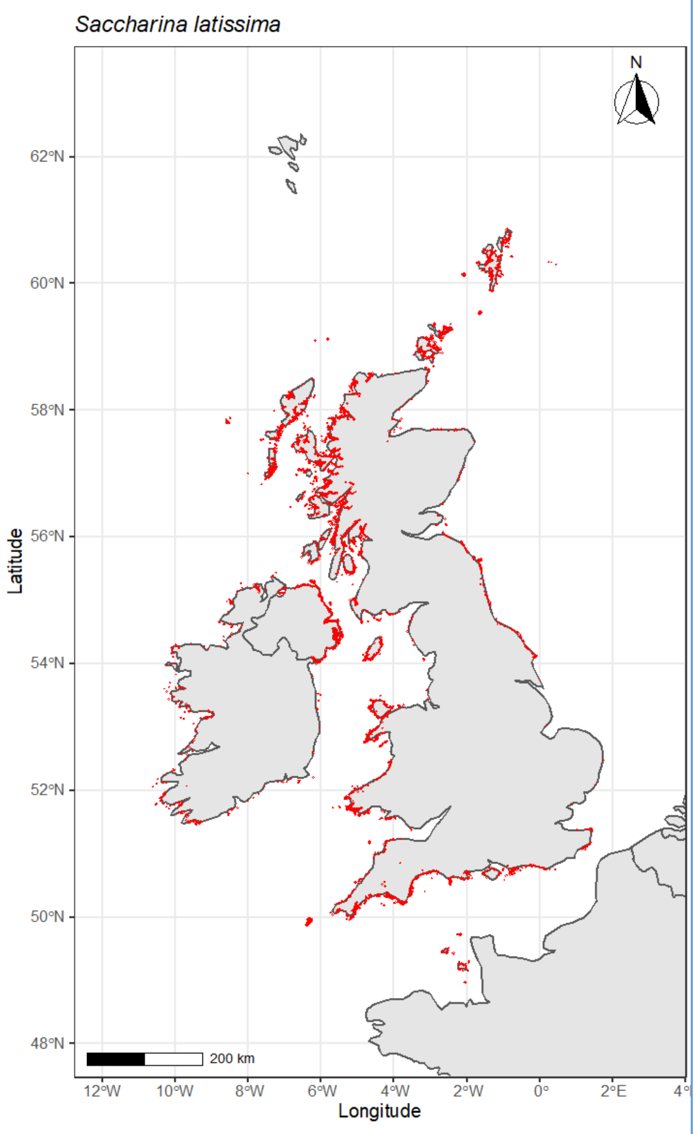

Figure 12. Distribution of *S. latissima* in the United Kingdom and Ireland. Occurrences of *S. latissima* plotted as red points. Spatial data of species occurrences obtained from NBN Atlas, DASSH and Marine Recorder databases.

***Saccharina latissima*: Growth, reproduction and seasonality**

S. latissima typically lives for 2 to 4 years, potentially reaching a length of 4 m, and grows quickly from late winter through early spring, reaching maximum biomass in May–July (Nielsen *et al.,* 2014). In late summer and autumn, growth rates slow and spores are released from autumn to winter (Parke, 1948; Lüning, 1979; Birket *et al.,* 1998). The species may occur as a short lived perennial or an annual opportunist (Borum *et al.,* 2002).

The overall length of the sporophyte may not change during the growing season due to erosion from the tips (Bolton & Lüning, 1982, Nielsen *et al.,* 2014) with growth occurring between March and May (Nielsen *et al.,* 2014). Growth of the blade has been measured at 1.1 cm day^{-1}, with a total length addition of ≥2.25 m year^{-1} (Birket *et al.,* 1998), although a maximum growth rate of up to 4.87 cm day^{-1} has been recorded. Growth then declines from June onwards and may cease completely in late summer (Sjøtun, 1993), with high levels of erosion from July to November (Nielsen *et al.,* 2014). At the end of summer, effort is shifted into increasing width, rather than length, of the blade, which may be a strategy to maximize the blade area for autumn and winter and therefore increase the amount of stored carbon available for plants at this time (Sjøtun, 1993).

S. latissima has a two-phase lifecycle, which alternates between a visible sporophyte (the seaweed 'plant' seen on the shore) and microscopic, filamentous gametophyte stages. Once mature at 8 to 15 months, specialised "sorus" tissue forms along the centre portion of the blade, from which large numbers (>1,000,000) of zoospores are released (Kain, 1979). This tissue may be present all year round, although October to April was the most frequent period of spore production in the British Isles (Parke, 1948). The zoospores remain in the water column for a period of 24 hours before settling onto the seabed (Birkett *et al.,* 1998b). These spores develop into gametophytes, which become fertile in under 10 days in optimal conditions. The gametophytes produce gametes (sperm and eggs) which fuse after fertilization, forming juvenile platelets (germlings) *in situ* of the female gametophyte. *S. latissima* recruits appear in late winter and early spring, peaking in December to January (Andersen *et al.,* 2011).

If conditions are not optimal, the gametophytes can develop vegetatively, forming an invisible "seed bank" in the understory, which may persist for several years (Van den Hoek *et al.,* 1995 cited in Sjøtun and Schoschina 2002). Fragments of damaged vegetative gametophytes may develop into separate gametophytes (only a few cells are required) hence reproductive potential may be increased. If optimal conditions return the gametophyte may become fertile and produce gametes. However, successful fertilization requires a high density of spore settlement (about 1 mm apart).

Temperature is a major factor affecting both reproductive activity (Bartsch *et al.,* 2013) and growth in *S. latissima*, with decreased growth rates evident above 16°C, and a 50-70% growth reduction at 20°C (Bolton and Lüning, 1982; Nepper-Davidsen *et al.*, 2019).

Growth increases with lower wave exposures, and was ~40% less at exposed locations compared to sheltered or moderately exposed locations, however more exposed coastal areas to limit biofouling, which can be beneficial for reproduction (Visch *et al.,* 2020a &b).

Saccharina latissima: Recruitment and recovery potential

S. latissima is an opportunistic seaweed exhibiting relatively fast growth rates and early maturation compared to other perennial species.

The species can rapidly recolonize cleared areas of the substratum. Kain (1975) recorded that *S. latissima* was abundant six months after substratum was cleared of all vegetation.

Without competition from other kelp species, *S. latissima* populations increase their biomass within two years, while its density decreases (Mikhaylova, 1999). Re-attachment of dislodged *S. latissima* may occur in certain conditions, with dislodged individuals growing new holdfasts to subsequently attach to the substratum (Burrows, 1958).

Phase shifts

Despite this, following the loss of *S. latissima* forests from Norway, there was a shift to an alternative stable state dominated by ephemeral algae (Moy and Christie 2012). While the drivers of this are unclear, and harvesting is not apparently a key factor (eutrophication and climate change are suggested), it is clear that recovery of *S. latissima* does not always occur.

Dispersal potential

Evidence on the spore dispersal of *S. latissima* is limited. The passive dispersal of spores is reliant on local current and wave mediated water movements (Cie & Edwards, 2011), and kelp larval dispersal varies with location and species. Kelp zoospores are expected to have a large dispersal range, however zoospore density and the rate of successful fertilization decreases exponentially with distance from the parental source (Fredriksen *et al.,* 1995). Hence, recruitment following disturbance will be influenced by the proximity of mature kelp beds producing viable zoospores to the disturbed area (Kain, 1979; Fredriksen *et al.,* 1995).

In conditions of low water movement, which occur in some *S. latissima* dominated biotopes, larval dispersal range is likely to be depressed, with the majority of recruitment occurring within the kelp bed. The reforestation of historic kelp beds off Norway indicate that natural recolonization was prevalent in the past (Moy and Christie, 2012). Andersen (2013) suggests that this, and other regional studies (see Andersen 2013 and the references herein) are illustrative of population connectivity and long distance dispersal in *S. latissima. S. latissima* exhibits a high degree of plasticity between populations with differing tolerance to a range of temperatures reported from population in Maine and New York in the USA (Gerard and Du Bois, 1988).

The recovery of *S. latissima* forests is thought to be impaired by epiphyte fouling (Andersen *et al.,* 2011). Uncontrolled grazing of kelps by herbivores, including sea urchins, may prevent kelp regeneration after harvesting. Removal of urchin predators through direct harvesting (e.g. of fin fish) or indirect elimination of the kelp canopy, can lead to an urchin population increase which, unchecked by predation may result in the formation of barrens and the loss of *S. latissima* biotopes (Bernstein *et al.,* 1981; Estes & Duggins 1995; Ling *et al.,* 2009). Heavy biofouling has been indicated to cause premature death and decreased reproductive output in *S. latissima* (Saier and Chapman, 2004, Andersen *et al.,* 2011). This indicates that a decrease in grazers which feed on these epibionts could be detrimental to recovery, especially in the light of future global sea temperature increases, which favour the growth of ephemeral algae (Andersen *et al.,* 2011).

Saccharina latissima: Ecological importance of species

Habitat provision

S. latissima can dominate areas subject to recurrent or intermittent disturbance or in areas where environmental conditions are unfavourable to other kelp species. As such, it represents an important habitat former in these areas.

Habitat provision- epiphytes

S. latissima epiphyte communities can include blue mussels (*Mytilus edulis*), sponges, bryozoans (*Membranipora membranacea, Electra pilosa,* and *Celleporella hyalina*), filamentous algae, and the vase tunicate (*Ciona intestinalis*). Although the presence of these epiphytes can increase drag forces leading to kelp mortality (Andersen *et al.,* 2011; Førde *et al.,* 2016).

The Blue rayed limpet *Patella pellucida* feeds directly on this kelp species, and the urchin *Echinus esculentus* preferentially feeds on bryozoan encrusted *S. latissima* over *Laminaria digitata* (Bonsdorff & Vahl, 1982).

The communities associated with cultivated *S. latissima* have been well studied, in efforts to reduce fouling and improve crop quality. However due to the differing structure of line-grown kelp holdfasts, and their location in the water column, caution should be exercised in generalising to wild populations (Walls *et al.,* 2017).

Habitat modification (including shading and moisture retention)

In a recent review, Teagle *et al.* (2017) outlined that kelp canopies alter light (Connell, 2003a), sedimentation (Connell, 2003b), physical abrasion (Irving and Connell, 2006), flow dynamics (Eckman *et al.,* 1989), substratum availability and condition (Christie *et al.,* 2007) and food quantity and quality (Krumhansl and Scheibling, 2012).

Bioremediation

Due to its ability to absorb extra nutrients *S. latissima* can be valuable in bioremediation of eutrophicated waters, (Bruhn *et al.,* 2016), with applications for mitigating the environmental impact of aquaculture through Integrated Multi-Trophic systems (Marinho *et al.,* 2015; Bruhn *et al.,* 2016; Gundersen *et al.,* 2016).

In a Swedish study, estimates of nutrient mitigation capacity indicated that under the best available conditions, *S. latissima* removes approximately 100 kg N and 1000 kg C $ha^{-1}yr^{-1}$, which is comparable to mussels at approximately 700 kg N and 6600 kg C $ha^{-1}yr^{-1}$ (Visch *et al.,* 2020).

Carbon capture, nutrient cycling and trophic transfer

Through erosion and shedding of portions of the blade, *S. latissima* may also play an important role in the export of detritus and carbon capture (Krumhansl and Scheibling, 2012).; Filbee-Dexter *et al.,* 2020). *S. latissima* has a clearly defined seasonal pattern of growth and decay that influences its nutrient and carbon cycling and the turnover of organic matter and structure of the habitat (Krumhansl and Scheibling, 2011).

Saccharina latissima: Ecological implications of harvesting

The recovery of *S. latissima* associated biota is likely to depend on the recovery of *S. latissima* and therefore likely to lag slightly after the initial stages of recovery by *S. latissima.*

Phase shifts

Following the loss of *S. latissima* forests from Norway, shifts to an alternative stable state have been documented, either dominated by ephemeral algae (Moy and Christie 2012) or resulting in the formation of “urchin barrens” (Hynes *et al.,* 2021), which support few other ecosystem functions and have low productivity (Christie *et al.,* 2009). While the drivers of these shifts are unclear, and harvesting is not apparently a key factor (eutrophication and climate change are suggested by Moy and Christie 2012), it is clear that overharvesting could have potentially drastic consequences.

Relevant studies or monitoring

See studies above on recovery following clearance.

Moy and Christie (2012) recorded a large scale decline of *S. latissima* on the Norwegian coast, discovered in 2002. They then conducted a large survey between 2004-2009 of 660 sites covering 34,000km of south and west Norway to assess the decline of *S. latissima* abundance and distribution (Moy & Christie, 2012). The survey indicated an 83% reduction of *S. latissima* forests across the south Norwegian region of Skagerrak. The west Norwegian coast was less affected, but *S. latissima* was either absent or very sparse at 38% of sites where it was expected to be abundant. At all sites where *S. latissima* was sparse a community of ephemeral macro-algae species was dominant and persisted throughout the study period (2004-2009). Bekby & Moy (2011) modelled the regional decline which indicated a decline of 50.7% of *S. latissima* from Skagerrak, Norway.

Approximately 50% of Europe’s *S. latissima* is found in Norway (Moy *et al.,* 2006), therefore, despite large discrepancies between the two estimates of *S. latissima* decline (50.7-83%) the results indicated a significant decline in *S. latissima* across the region. Moy & Christie (2012) suggested the ephemeral filamentous macroalgae communities represented a stable state shift that had persisted throughout the study period (2004-2009). Although no measurements were made, they suggested that the decline was due to low tidal movement and wave action in the worst affected areas combined with the impacts of dense human populations and increased land run-off multiple stressors such as eutrophication, increasing regional temperature, increased siltation and overfishing (with knock on impacts on grazing species) may also be acting synergistically to cause the observed habitat shift.

Species specific management approaches

- Harvest during the peak growth season in spring and summer

- Cut the frond above the meristem; Ireland: cut the frond at least 20 cm above the stipe (Edwards *et al.,* 2011)
- Avoid harvest during reproductive season in autumn and winter / Avoid harvesting fertile individuals

Access to *S. latissima* from the shore is limited by the tidal window, which could act as a buffer against excessive harvesting. However, should use of snorkel or SCUBA increase, harvesting pressure would increase correspondingly, potentially requiring additional management.

Although removal of the whole alga may occur if appropriate guidance is not followed, no evidence of this practice was found from businesses harvesting commercially. Hand harvest in England and Wales is thought to mirror that in Scotland, in which the stipe and lower portion of the blade are left intact and juvenile plants are avoided (Scottish Government 2016).

Saccharina latissima: Knowledge Gaps

- Baseline standing stock data is absent for England and Wales
- Sustainable harvest volumes are unknown
- Dispersal potential
- Genetic structure of population
- Cumulative stress responses – e.g. from rising temperature, increased severity and frequency of storms and harvesting

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Appendix 13 *Saccorhiza polyschides*

Names

Saccorhiza polyschides

Furbellow, Sea Hedgehog, Irish: Claíomh, Madra, Clabhthai

Saccorhiza polyschides: harvesting

Currently *S. polyschides* is hand harvested in England and Wales to a limited extent for food purposes, where it is usually included as a mixed, flaked product. Although *S. polyschides* is edible (Sánchez-Machado *et al.,* 2004), it is not currently listed as a novel food by the EU, nor is it commercially harvested for alginate production at present, but it may be of interest in the future because of its fast growth rate.

This species is not listed as harvested in Scotland (Scottish Government 2016) nor it is included in guidance produced by Natural England (Bailey & Owen 2014), however harvest guidance exists for Ireland (Edwards *et al.,* 2012) and it has been cultivated during trials in Scotland and England.

Werner and Kraan (2004) state that *S. polyschides* has lower economic value than for example *Laminaria digitata*, however it has been identified as having potential for use as fertilizer (Soares *et al.,* 2020), for alginate and other industrial biopolymers (Silva *et al.,* 2015) in food, pharmaceutical, and cosmetic industries (Pereira *et al.,* 2013). A recent paper also outlined the use of an extract of *S. polyschides* in the synthesis of gold and silver nanoparticles (González-Ballesteros *et al.,* 2021).

Saccorhiza polyschides: Natural range / distribution

S. polyschides colonizes hard substrata from the low water mark to a depth of 35 m. It normally attaches to bedrock, boulders and cobbles but is occasionally found loose-lying attached to small stones or shells. It is found in conditions from very exposed to ultra sheltered, and forms dense stands in sheltered areas and mixed stands with *Laminaria* spp. in medium to exposed habitats (Hawkins and Harkin 1985; Smale *et al.,* 2017). It can tolerate a wide range of tidal flows, from negligible to very strong > 6 knots (>3 m/sec.) currents. Found only in fully saline conditions, it is tolerant of some sand scour.

S. polyschides has a wide geographic distribution and can tolerate a wide range of temperatures. Sporophyte growth can occur from 3-24°C and gametophyte development from 5-25°C (Norton, 1977). Fernández (2011) however suggested that summer temperatures of >20°C sustained for longer than a period of 30 days may inhibit development and recruitment. The species appears to have benefitted from recent increases in sea temperature in Brittany, where it quickly colonises disturbed areas in the *Laminaria digitata* canopy (Arzel, 1998 In Werner and Kraan 2004). Increasing sea temperature and resulting changes in the *Laminaria* spp. community might benefit *S. polyschides* in Southwest England (Birchenough and Bremner 2010; Smale *et al.,* 2013;

Yesson *et al.,* 2015). An increase of *S. polyschides* populations is predicted for northern regions of the NE Atlantic under continued warming (Assis *et al.,* 2018).

The shape of the frond varies with the degree and nature of water movement. In sites of low water current plants produce broad undivided fronds, while those in areas of strong currents have long deeply divided fronds. Plants from wave exposed locations have short fronds divided into few sections (Norton, 1978).

It's distribution includes the all coasts of Britain and Ireland (see Figure 13), but is absent from Northumberland to the Solent. More widely, it extends from Morocco northwards along the European coastline, with the most northerly recorded location at Rorvik, Norway. It has also been reported in the Eastern Mediterranean extending to the Greek coastline, and Italy (Lüning, 1990).

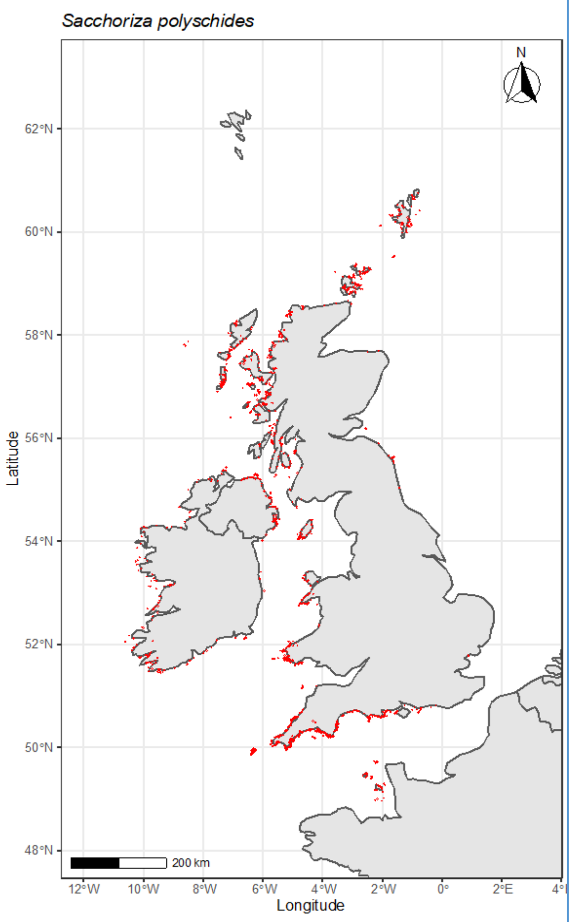

Figure 13. Distribution of *S. polyschides* in the United Kingdom and Ireland. Occurrences of *S. polyschides* plotted as red points. Spatial data of species occurrences obtained from NBN Atlas, DASSH and Marine Recorder databases.

Saccorhiza polyschides: Growth, reproduction and seasonality

The species is an opportunistic pseudo-annual, and very fast growing. Growing up to 4 m long, plants typically have a lifespan of 8-10 months. However, those which do not reach maturity within the first growth season can overwinter and live for 12-18 months (Birket *et al.,* 1998; Fernández, 2011; Norton and Burrows 1969). Age at maturity is between 8 – 14 months.

The large seaweed ‘plants’ (sporophytes) are visible on the shore from March until winter. Growing rapidly through spring and early summer at a rate of ≤6.2 cm per week or up to 2 m a month (Norton, 1970; Fernández, 2011), the onset of maturity triggers senescence, with growth ceasing and the frond beginning to erode. Through late summer - autumn spores are released and the frond and stipe continue to decay, leaving behind the bulbous holdfast, which remains on the shore until it is detached by winter storms (Birket *et al.,* 1998; Fernández, 2011).

S. polyschides exhibits a bi-phasic lifecycle, alternating between a visible sporophyte (sometimes referred to as a “plant”), and a microscopic, filamentous gametophyte stage. Fertile “sorus” material is formed on the stipe (most densely on the frilly “sporophyll” structures) and blade, from which large numbers (>1,000,000) of zoospores are released. Most (75%) zoospores remain in the water column for 24 hours and may be transported at least 200 m from the parent before settling onto the seabed. There they develop into either male or female gametophytes, with the male gametes fertilizing the female within 10 days in optimal conditions, and a juvenile plantlet/germling developing in situ. Gametophytes can remain in vegetative reproduction until optimal conditions prompt fertility, forming a miscroscopic “seed bank” from which visible plants can develop.

The unusual holdfast, or “bulb” of *S. polyschides* is formed from a hollow bulbous growth above the sapling holdfast which expands to overwhelm it, sending out secondary haptera to attach to the substratum.

Recruitment and recovery potential

Due to the fast growing, opportunistic, annual nature of this species, dense stands can develop rapidly on freshly cleared areas of rock, becoming established in less than a year, providing that sources of spores are available from nearby populations.

Experiments have shown that *S. polyschides* colonizes cleared areas of the substratum within 26 weeks. However, if clearance takes place in August, when no spores of the species are released, the substratum may become colonized by red algae potentially blocking colonization by *S. polyschides* (Kain, 1975).

In Brittany, *S. polyschides* quickly colonises disturbed areas in the *Laminaria digitata* canopy (Arzel, 1998 In Werner and Kraan 2004), and appears to benefit from harvest induced disturbance to the *Laminaria* canopy in combination with increases in sea temperature.

Kelp zoospores are expected to have a large dispersal range, but for fertilisation to be successful, gametophytes must settle at a high density (within 1 mm of each other) (Reed, 1990; cited in Birkett *et al.,*1998b). Therefor recruitment is influenced by the proximity of mature kelp beds producing viable zoospores (Kain, 1979; Fredriksen *et al.,*1995).

Saccorhiza polyschides: Ecological importance of species

As primary producers and ecosystem engineers, kelps provide a variety of ecosystem services (Smale *et al.,* 2013). They support biodiversity by provision of a complex three-dimensional habitat structure, deliver high primary productivity, and amplified secondary productivity (Teagle *et al.,* 2017; Smale *et al.,* 2013). They play in important role in nutrient cycling, carbon capture and transfer, and by dampening wave energy provide coastal defence. They are also valuable for water purification (Vasquez *et al.,* 2014) and carbon storage (Gundersen *et al.,* 2010).

Habitat modification (including shading and moisture retention)

In a recent review, Teagle *et al.* (2018) outlined that kelp canopies alter light, sedimentation (Connell, 2003), physical abrasion (Irving and Connell, 2006), flow dynamics (Eckman *et al.,* 1989), substratum availability and condition (Christie *et al.,* 2007) and food quantity and quality (Krumhansl and Scheibling, 2012).

Habitat provision

Relative to Laminarial kelps, *S. polyschides* is fast growing and short lived, which will limit the opportunity for a diversity climax community to develop in association with this kelp. The gradual loss of plant parts over winter, will temporally limit the habitat value of *S. polyschides.* Despite this, its ecological importance is still considerable. Biotopes characterised by *S. polyschides* are generally those in which disturbance events prevent other Laminarial kelps from becoming dominant through the process of succession. These include "*S. polyschides* and other opportunistic kelps on disturbed sublittoral fringe rock", found mainly in the southwest and west England, and "Saccharina latissima and/or *S. polyschides* on exposed infralittoral rock", which result from disturbance by storms, when loose sediment and even cobbles or boulders are mobilised, scouring most seaweeds and animals from the surrounding rock. As a result of the transient nature of these biotopes, their composition varies both temporally and regionally.

Habitat provision- epiphytes

The different parts of the plant (blade, stipe / sporophyll and holdfast) support different associated species, with a total of 89 animal species recorded (Norton, 1971). Although no species are exclusively reliant on *S. polyschides,* Norton (1971) found that several animal species were much more abundant on *S. polyschides* than on other laminarians:

Habitat provision-holdfasts

The unusual bulbus holdfast of this species is large and hollow, creating a high volume micro-habitat which is quite structurally different from the Laminarian kelp species. *S. polyschides* holdfasts contain a lower diversity and abundance of species in comparison *Laminaria hyperborea* but consist of larger animals, including several predatory fish and crustaceans that are typically absent from *L. hyperborea* (McKenzie & Moore, 1981; Salland and Smale 2021a,b).

The holdfasts of *S. polyschides* are known to shelter large animals such as large polychaetes, squat lobsters and fish which shelter inside the bulbous holdfast, while amphipods, brittle stars and polychaetes occur in the space between the base of the bulb and the rock surface to which it is attached (McKenzie & Moore, 1981; Teagle *et al.,* 2018). The composition of the epifauna and epiflora varies with environment factors, particularly the degree of water movement and of siltation (Ebling *et al.,* 1948; Norton, 1971). The fronds are grazed by urchins such as *Echinus esculentus* and *Paracentrotus lividus*, and the blue-rayed limpet *Patella pellucida.*

Habitat provision- understorey

S, latissima dominated biotopes support an understorey community characterized by scour tolerant or ephemeral red seaweeds. Foliose red seaweeds, green seaweed *Ulva* spp and brown seaweed also occur.

Faunal diversity and abundance are also generally low and sparse, typically limited to encrusting bryozoans and/or sponges, such as *Halichondria panicea* and the gastropod *Gibbula cineraria*. (Information from Connor *et al.,* 2004). The kelp provides a habitat for a variety of encrusting bryozoans and the holdfasts may be colonized by mobile species including polychaete worms, crustaceans and prosobranch molluscs.

Kelps may provide local shelter for small fish such as the two-spotted goby *Gobiusculus flavescens.*

Physical and chemical

Due to its rapid growth and high productivity this species is likely to play an important role in the turnover of organic matter, export of detritus, carbon capture, and nutrient cycling however no literature specific to *S. polyschides* was found within the time frame of the review.

Saccorhiza polyschides: Ecological implications of harvesting

No evidence was found specifically on the impacts of hand harvesting this species. As it is naturally short lived and opportunistic, impacts are expected to be relatively minimal in comparison to natural population fluctuations. Harvesting following guidance to remove part of the blade, leaving the meristem at the base from which growth occurs, are likely to result in rapid recovery either by continued blade growth or new recruits from the understory. The time frame of the impacts from loss of the canopy will be reduced if harvesting takes place in spring or early summer.

If the canopy were removed, the red seaweeds understorey community may become bleached, and/or perish (Hawkins & Harkin, 1985), leading to reductions in biodiversity.

Relevant studies or monitoring

See above.

Species specific management approaches

- Leave the lower 20 cm of the frond (above the meristem) (Edwards *et al.,* 2012)
- Avoid harvesting fertile material

Based on the evidence above provided by Kain (1975), management of this species should not allow for clearance of entire plants from areas in August, as the opportunity for reproduction will be lost. However, if harvesting were to remove the blade only, leaving the stipe and sporophylls, then the impact on reproductive output would be minimised.

Recovery time will be reduced if harvesting takes place in spring or early summer, during the peak growth season. Harvesting during autumn and winter will be naturally restricted by the absence of all but the bulb from the coasts of England and Wales.

Access to this seaweed resource from the shore is limited by the tidal window, which could act as a buffer against excessive harvesting. However, should use of snorkel or SCUBA increase, harvesting pressure would increase correspondingly, potentially requiring additional management.

Saccorhiza polyschides: Knowledge Gaps

- Baseline standing stock biomass
- Sustainable harvesting volumes

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Appendix 14 *Ulva spp.*

Names

Ulva spp.

Sea lettuce, Green laver, Glasán

Ulva spp: harvesting

Ulva spp. are hand harvested both commercially and recreationally in England and Wales by picking or using scissors.

Ulva spp. are collected for use in pharmaceuticals and food. It can be eaten fresh or dried, and may be eaten as a salads or garnish. Other uses include as a manure (from drift *Ulva* in Brazil) or fertilizer, source of bioethanol, or substrate for paper production (Konur, 2020)

A recent project hosted at Aberystwyth is exploring the potential for harvesting *Ulva* blooms as a mechanism for bioremediation (Oliver pers. comm)

Ulva spp: Natural range / distribution

Ulva spp. are found at all levels of the shore, where it grows attached to various substrates including rock, shells, pebbles and sometimes other algae via a small hold-fast. They are opportunistic, with broad tolerances for a wide range of conditions (Vermaat & Sand-Jensen, 1987). It can grow on a wide variety of sediment types, shore heights, wave exposures and salinity regimes. The genus also forms dense cover in rockpools, particularly on the very upper reaches of the shore. Found intertidally and into the shallow sublittoral, it is absent from only the most exposed rocky shores.

In very sheltered conditions, plants that have become detached from the substrate can continue to grow, forming extensive floating communities. The plant tolerates brackish conditions, can be found on suitable substrata in estuaries, and can proliferate forming “blooms” in nutrient enriched conditions. Examples of these in Wales and England Wales include Millford Haven and the Solent respectively.

Ulva spp. have a wide thermal tolerance and are recorded from all British coasts (see Figure 14), from France to Denmark. The *Ulva* genus is globally distrubuted, although the species are likely to differ regionally.

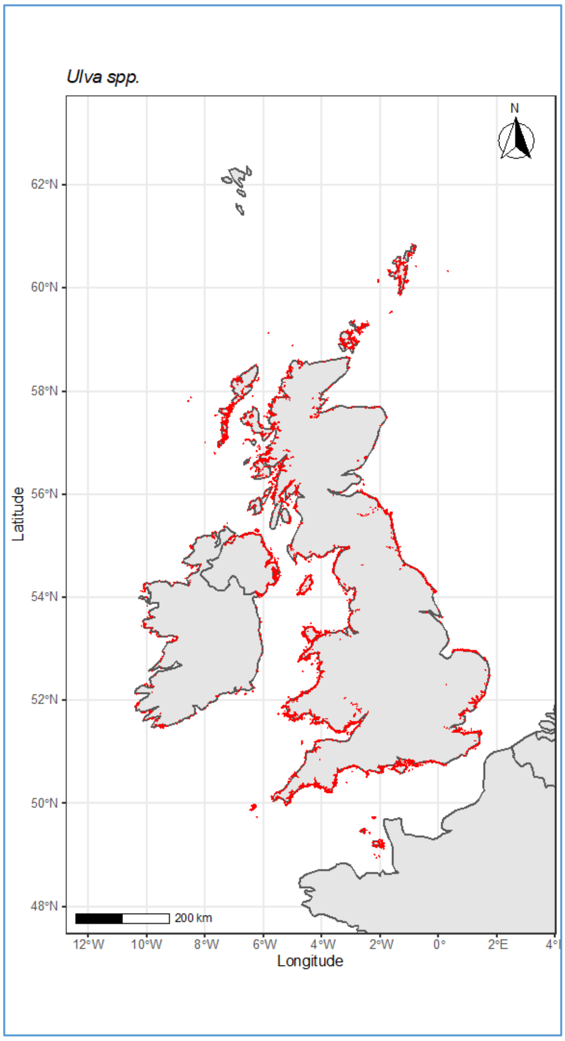

Figure 14. Distribution of Ulva spp. in the United Kingdom and Ireland. Occurrences of Ulva spp. plotted as red points. Spatial data of species occurrences obtained from NBN Atlas, DASSH and Marine Recorder databases.

Ulva spp: Growth, reproduction and seasonality

In England and Wales, *Ulva spp.* includes the flat, sheet-like foliose from (*Ulva lactuca*) and the hollow tubular form (*U. lactuca* previously *Enteromorpha intestinalis*). The genus *Ulva* currently contains numerous species (Guiry & Guiry, 2015), and is accepted name for *Enteromorpha* and *Chloropclta* which were previously considered as separate genus (Hayden *et al.,* 2003). Taxonomy of the genus is rapidly evolving and identification to species level can be problematic, in some instances species can only be distinguished by experts or by genetic analysis.

The sheet-like form generally referred to as Sea Lettuce (*U. lactuca*) grows attached to rocks directly by its holdfast. It can reach 45 centimetres long and 30 centimetres wide, although it is typically much smaller. *Ulva spp.* are opportunistic and are able to rapidly colonize newly created space. The life history characteristics that support this opportunism are high growth and reproduction rates.

During the spring and summer, particularly in nutrient enriched conditions with freshwater influence, *Ulva spp.* can grow rapidly to form extensive blooms. While potentially harvested to mitigate eutrophication, these blooms are unlikely to be utilised for human consumption due to the risk of contamination. *U. lactuca*, Sea lettuce, generally occurring on the rocky shore is more likely to be harvested for food purposes.

Reproduction

Ulva spp. have extended reproduction periods (Smith, 1947). The short lived plants reach maturity at a certain stage of development rather than relying on an environmental trigger. For *U. intestinalis,* maturity is reached in as little as a few weeks. *U. intestinalis* can be found in reproductive condition at all times of the year, but maximum development and reproduction occur during the summer months especially towards the northern end of the distribution of the species (Burrows, 1991).

The life history consists of an alternation between haploid gametophytic and diploid sporophytic generations, but can be modified by environmental conditions (Burrows, 1959; Moss & Marsland, 1976; Reed & Russell, 1979). The gametophytes of *Ulva* produce enormous numbers of mobile gametes which cluster and fuse to produce a sporophyte (diploid zygote). The sporophyte matures and produces large numbers of zoospores that mature as gametophytes, and the cycle is repeated.

The zoospores and gametes (collectively called swarmers) are released into the water column in high numbers, and can disperse over large distances. They may be released in such quantities into rock pools or slack water that the water mass is coloured green (Little & Kitching, 1996). *Ulva spp.* can form the swarmers from normal frond cells that are transformed into reproductive tissue rather than having to produce specialised reproductive structures (Lersten & Voth, 1960), so that a significant portion of the macroalga's biomass is allocated to the formation of zoospores and gametes (Niesenbaum, 1988).

Swarmers are often released in relation to tidal cycles, with the release being triggered by the incoming tide as it wets the frond. However, the degree of release is usually related to the stage of the spring/neap tidal cycle, so allowing regular periodicity and synchronization of reproduction (Little & Kitching, 1996). Christie & Evans (1962) found that swarmer release of *U. intestinalis* from the Menai Straits, Wales, peaked just before the highest tides of each neap-spring cycle. Mobility of *U. intestinalis* swarmers can be maintained for as long as 8 days (Jones & Babb, 1968), allowing them to disperse over large distances e.g. 35 km (Amsler & Searles, 1980).

Recruitment and recovery potential

Both *U.* (was *Enteromorpha*) *intestinalis* and *U. lactuca* are classified as opportunistic species that are able to rapidly colonize newly created gaps across a range of habitats.

Recovery rates are rapid due to the supply of swarmers, which can disperse over distances of at least 35 km in vast numbers (Amsler & Searles 1980; Niesenbaum, 1988). *U. intestinalis* is amongst the first multicellular algae to appear on substrata that have been cleared following a disturbance, e.g. following the Torrey Canyon oil spill in March 1967, species of the genus *Ulva* rapidly recruited to areas where oil had killed the herbivores that usually grazed on them, so that a rapid greening of the rocks (owing to a thick coating of Ulva *spp.*) was apparent by mid-May (Smith, 1968). The rapid recruitment of Ulva *spp.* to areas cleared of herbivorous grazers was also demonstrated by Kitching & Thain (1983). Following the removal of the urchin *Paracentrotus lividus* from areas of Lough Hyne, Ireland, *Ulva* grew over the cleared area and reached 100% coverage within one year.

Available evidence indicates that these species will recover rapidly, even following removal of the whole alga from large areas.

Ulva spp: Ecological importance of species

Habitat and food provision

Ulva spp. is grazed by a variety of species, including littorind snails, which can prevent blooms of Ulva *spp.* from forming (Robles, 1982, Albrecht, 1998). The genus is also eaten at low tide by Brent Geese (Light-bellied Brent). In shallow upper shore rock pools it can form dense coverage. In these habitats it provides shelter for the orange harpacticoid copepod, *Tigriopus brevicornis*, and the chironomid larva, *Halocladius fucicola* (McAllen, 1999). *U. intestinalis* is often the only seaweed found in rockpools at the very upper shore, and the copepod and chironomid species utilize the hollow frond as a moist refuge from desiccation when the rock pools completely dry out.

Primary production and physical processes

Information specific to *Ulva spp.* was not found, but all algae contribute to primary production, and accumulations of algal debris are likely in habitats such as the high shore rock pools of some *Ulva* biotopes, where such detrital material contributes to overall productivity.

Ulva spp: Ecological implications of harvesting

No literature was found specific to this genus. As these species are fast growing, opportunistic, and ephemeral, hand harvesting is unlikely to have a substantial impact on populations of the algae themselves.

However, harvesting activity, particularly in areas like estuaries where large blooms are known to occur, may disturb wading birds and access may involve trampling over saltmarsh habitats.

Relevant studies or monitoring

Ecological surveys to monitor the impacts of harvesting are necessary.

Species specific management approaches

- Harvest during the season or rapid growth (spring and summer)

Ulva spp: Knowledge Gaps

- Standing stock biomass for England and Wales are unknown, and are likely to fluctuate widely due to the opportunistic nature of summer blooms
- No evidence on sustainable harvesting practices or volumes was found

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Appendix 15 Recovery rates of harvested species

The recovery rates presented below (Table 15.1) have been assessed using the categories used for the Marine Life Information Network (www.marlin.ac.uk) standard recovery assessments used for sensitivity assessments. ‘Full recovery’ is envisaged as a return to the state of the habitat that existed prior to impact. However, this does not necessarily mean that every component species has returned to its prior condition, abundance or extent but that the relevant functional components are present and the habitat is structurally and functionally recognizable as the initial habitat. Recovery assessments are based on the assumption that best practice methods are used and relate to a single harvesting event. i.e. partial frond harvest is based on harvesting following NRW guidance on the proportion of the frond remaining (see section 4); recovery from total clearance is based on small (2m^2) areas with adult source populations near by to supply propagules. Recovery rates following more intensive, repeated or extensive harvesting, for example where large areas of shore are denuded of seaweed cover, will be slower than those presented in the table. Evidence indicates the level of confidence based on the available literature.

Table 10. Marine Life Information Network (MarLIN) recovery categories.

Very low	Negligible or prolonged recovery possible; at least 25 years to recover structure and function
Low	Full recovery within 10-25 years
Medium	Full recovery within 2-10 years
High	Full recovery within <2 years

Table 11. Recovery rates of harvested species.

Scientific name	**Growth rate / season**	**Max. typical size**	**Lifespan**	**Age at maturity**	**Reproduction season**	**Recovery from partial frond harvest**	**Recovery from total clearance**	**Evidence / confidence**
A. esculenta	Perennial. From base of blade. Peaks in spring (April – May) at 20 cm / month; erodes from June – winter leaving only midrib stalk	90cm	4-7 years	8-14 months	Nov – March, spores released from sporophylls at the base of the plant. Recruits appear in spring	High (in spring growth season)	High (1-9 months in small 2m^2 areas)	Low
L. digitata	Perennial. From base of blade. Peaks in winter (Feb – July). Slower in summer (Aug – Jan). Mean growth rate of 1.3 cm / day during max growth season	1-3m	4-6 years	18-20 months	Sorus material forms on blade year round, peaking in July-Aug and Nov-Dec. Recruits appear year round peaking in spring and autumn	High (in winter-spring growth season); canopy recovery in 18-20 months	High (2 years in small areas). Spore dispersal potential is 200-600m	High – extensive primary literature. Although focus is skewed towards mechanised harvest in Europe

Scientific name	**Growth rate / season**	**Max. typical size**	**Lifespan**	**Age at maturity**	**Reproduction season**	**Recovery from partial frond harvest**	**Recovery from total clearance**	**Evidence / confidence**
L. hyperborea	Perennial growing from the base of the blade. Peaks in winter (Nov – June), slower in summer. Growth rate of 0.94 cm / day during max growth season	<3.5m	11-20 years	2-6 years	Sorus material forms on blade year round, peaking in winter (Sept – April). Recruits appear year round peaking in spring	High (in winter-spring growth season)	Medium (2-6 years) Spore dispersal potential is 5km	High – extensive primary literature, although focus is skewed towards mechanised harvest in Europe
S. latissima	Annual or short lived perennial growing from the base of the blade. Peak growth in late winter – spring (1.1 cm / day). Slower in summer.	<4m	2-4 years	8-15 months	Sorus material forms along the centre of the blade year round, peaking Oct – April. Recruits appear in winter - spring peaking in Dec and Jan	High (in winter-spring growth season)	High (6 months in small 2m^2 areas); limited evidence of spore dispersal, which is dependent on currents	High – extensive primary literature
S. polyschides	Annual. Peak growth in late	<4m	8-18 months	8-14 months	Sorus material forms on the	High (in spring	High providing	Medium (evidence

Scientific name	**Growth rate / season**	**Max. typical size**	**Lifespan**	**Age at maturity**	**Reproduction season**	**Recovery from partial frond harvest**	**Recovery from total clearance**	**Evidence / confidence**
	spring (6.2cm/week), senescing from mid-summer, absent by late winter				sporophylls and stipe, from which spores are released in summer-autumn. Recruits appear year round, peaking around June.	growth season)	source population are nearby, although highly seasonally dependant	for dispersal potential inferred from other kelp species)
A. nodosum	Perennial growing from the tip of the blade or regenerating from the base. Growth rate increases with age from 0.2 cm/year in the first year to 8-15cm/year when mature. Growth peaks in	0.5-2m	10-20 years per frond; >60 years per clump	>5 years	Vegetative reproduction from basal shoots is more common than sexual reproduction, which peaks in March-April.	Medium (3-5 years if 15-20cm left intact), dependant on cutting height – faster when more frond length is left to regenerate.	Low (12 years) dispersal is low and early mortality very high, with populations maintained by vegetative growth from	High – extensive literature relating to harvesting in the UK

Scientific name	Growth rate / season	Max. typical size	Lifespan	Age at maturity	Reproduction season	Recovery from partial frond harvest	Recovery from total clearance	Evidence / confidence
	spring/summer, slowing in autumn/winter						the holdfast.	
H. elongata	Bi-annual. The 'button' grows rapidly in diameter during the spring in its first year, producing reproductive 'straps' in autumn which grow rapidly in Feb-May the following year	2m	2-3 years	9-14 months	Reproductive 'straps' release gametes from June-winter, recruits appear from March	High (during spring peak growth) / 12 months provided that the 'buttons' remain	High (1-2 years) provided that source population are near by, although varies seasonally. Low dispersal potential.	Low – limited literature from Europe
F. serratus	Short lived perennial, growth occurs from the tips. Peak growth is spring-early summer, at	70cm	2-5 years	1-2 years	Gametes are released from the tips of the frond between late spring-autumn, peaking Aug-Oct	High (18 months to 2 years)	High (1 year in small 2m^2 plots); dispersal is limited to 0.3-0.3km	Medium

Scientific name	**Growth rate / season**	**Max. typical size**	**Lifespan**	**Age at maturity**	**Reproduction season**	**Recovery from partial frond harvest**	**Recovery from total clearance**	**Evidence / confidence**
	rates of 4-12 cm/year						from parent plant	
F. vesiculosus	Short lived perennial, growth occurs from the tips and peaks in summer. Peak growth rates are 0.14 cm/day	20cm – 2m	2-5 years	1-2 years	Gametes are released from the tips of the frond between winter – late summer, peaking May-June	High (12 months)	High (3-21 months) provided that source populations are near by; dispersal is limited	High relating to British or European populations
C. crispus	Perennial regenerates from the base. Growth peaks from May – Nov at rates of 0.33-0.37 mm/day	22cm	2-6 years	>2 years	Fertile material forms across the blade year round, peaking in winter.	High (6-18 months) depending on season and length of frond remaining	Medium (2-10 years)	High relating to British or European populations
M. stellatus	Perennial regenerates from the base. Growth peaks in spring-summer	17cm	Presumed 2-5 years	Presumed 1-2 years	Fertile material forms across the tips of blades	Presumed high (18 months) depending on season and length	Presumed medium (2-10 years)	Low – inferred from *C. crispus*

Scientific name	**Growth rate / season**	**Max. typical size**	**Lifespan**	**Age at maturity**	**Reproduction season**	**Recovery from partial frond harvest**	**Recovery from total clearance**	**Evidence / confidence**
	and slows in winter					of frond remaining		
P. palmata	Perennial. New tissue grows from tips, may annually regenerate from the base. Growth peaks from March – Aug and erosion occurs from Aug-March.	20-50cm	Unknown	Male: 9-12 months; Female: few days	Fertile material forms across the blade	High in spring and summer growing season	High (5 months over small 2m^2 plots), although recruitment is likely to be limited to 10's of meters	Low
Porphyra spp.	Annual. Dynamic and variable, peaking in spring	20cm	Unknown, likely <1 year	45 days	Reproductive material develops all year round, spore release peaks in spring and autumn. Recruits typically	High (2 months during the peak growing season)	High (2 months during the peak growing season)	Low

Scientific name	**Growth rate / season**	**Max. typical size**	**Lifespan**	**Age at maturity**	**Reproduction season**	**Recovery from partial frond harvest**	**Recovery from total clearance**	**Evidence / confidence**
					appear in winter.			
Ulva spp	Annual. Peak growth in spring and summer	45cm	Unknown, likely <1 year	Few weeks	Year round, peaking in summer. Blade tissue is transformed into reproductive tissue.	High (2 weeks during the peak growing season)	High (<1year) with large dispersal potential	Medium

Sources: MarLIN; Burrows *et al.,* 2018, Edwards *et al.,* 2012

Appendix 16 Review of current guidance

Table 12 Review of current guidance outlining knowledge gaps.

Reference	**Region**	**Species**	**Sustainable harvesting advice**	**Knowledge gaps**
NRW, 2018	Wales	Attached and drift weeds: Various: Green, red and brown, advice per species	Ensure the holdfast and some of the blade is left intact for re-growth. Harvest sparsely, max: one third of mature population. Rotate harvesting areas Limit harvesting drift weed during Oct-March due to overwintering birds	None stated
Scottish Government, 2016	Scotland	Various species: Includes regeneration periods	Harvesting of maerl should be prohibited, as well as seagrass due to small beds Only harvest in areas where it is sustainable Leave holdfast attached Cut well away from point of growth Remove less than 1/3 of each plant Harvest small % standing stock Rotate	No information available on what would be considered a significant volume of removal for small red and green seaweeds

			Prior to harvesting undertake assessment of stock and biomass, estimate % cover	
Morrison, 2018	Scotland	Kelps	Cut fronds well above the point of growth and always leave the holdfast attached; Harvest sparsely. Take less than one third of each plant to allow for regrowth; Harvest during the active growth season and after reproduction if possible; Avoid denuding entire patches of one species in one growth season; Rotate harvesting areas to allow ample time for recovery; Biosecurity advice to manage risk of spread of invasive non-native species. Of 20Mt *L. hyperborea* - 6.5 Mt predicted to be in harvestable area where biomass exceeds 5 kg/m^2	None stated
O'Hanlon and O'Hanlon 2018	Ireland	*F. serratus,* *F. vesiculosus*	Holdfast remains intact Rotation between 3 harvest zones allows for 3-year regrowth period between harvests Standing biomass of the 2 target seaweeds is 76.3 tones Seaweed gathered in 40kg sacks and 1-2 sacks at low tide period	None stated

			Annual harvest quantity = 2,000kg (2 tones wet weight, 50 sacks over the year, ~1 sack a week). *F. vesiculosus* recruits readily to cleared areas of the shore although full recovery may take 1-3 years *F. serratus* recovery will probably have occurred after a year	
McLaughlin, 2006	N. Ireland	Various Attached and drift weeds	Authors will support environmentally sensitive methods of seaweed harvesting, and promotes the retention of drift weed on beaches	Limited information on the biomass, distribution and productivity of living seaweed and drift weed around the Northern Ireland coastline Lack of specific information on the carrying capacity of marine ecosystems to support seaweed harvesting and mariculture Direct and indirect effects of harvesting on biodiversity and coastal processes in N. Ireland
Government Jersey, 2019	Jersey	Various	Harvesting of attached seaweed will only be permitted by non-mechanized means	None stated

			Daily bag limits would for the majority of Red and Brown seaweeds be set at 5kg for recreational and 10kg for commercial	
Burrows *et al.,* 2018	Scotland	Kelp	Proportion of standing stock advised for harvest varies by site from 10-35%, e.g. 6.5 Mt of *L. hyperborea* Five year harvesting rotation per area Harvesting in strips one nautical mile wide such that no strip borders one that has been previously harvested Boundary of the management area should be approximately 33x larger than the area estimated to contain the targeted total yield of kelp Need pre harvest assessments and monitoring including for recovery of associated biota	Impacts of harvesting kelp on commercial fish populations, no measurable impact from harvesting activities on stocks
Burrows *et al.,* 2010	Scotland	*A. nodosum*	Outer Hebrides was estimated at 170,500 tons of which 15,000-25,000 tones could be sustainably harvested annually	NA

Appendix 17 Confidence and likelihood of uptake of guidance measures

Table 13 Summary table of confidence in guidance measures, supporting rationale / likelihood of uptake summary table, structured by measure and species group. Based on guidance from Natural England and Natural Resources Wales Codes of Conduct (Bailey & Owen 2014; NRW 2018). Likelihood of uptake is based on authors opinion. Further supporting information for each species is provided in the Appendices 1-15.

Measures	**Species**	**Confidence in measure**	**Rational / Likelihood of uptake**
Avoid harvesting fertile material and harvest after the reproductive season	Kelps	High – repeated throughout guidance (intuitive precautionary advice, however limited primary literature was found)	*A. esculenta* and *S. polyschides*: the reproductive sporophylls are at the base of the stipe, so are easily avoided and are unlikely to be targeted by hand harvesting. Laminarian kelps: this strategy requires seasonal harvesting at times outside of the reproductive period, as the reproductive sorus material forms on the blade which is likely to be harvested. Confidence that recreational harvesters will have access to this understanding is low.
Avoid harvesting fertile material and harvest after the reproductive season	Browns	High – repeated throughout guidance (intuitive precautionary advice, however limited primary literature was found)	*F. vesiculosus* and *F. serratus* avoid taking only large mature individuals, which contribute most to the reproductive output. For *H. elongata* take only one of the two reproductive fronds. Likely to be effective if enforced, however confidence that recreational harvesters will have access to this understanding was low.
Avoid harvesting fertile material and harvest after the reproductive season	Reds	Medium – consistent through most guidance and supported by primary literature	Avoiding harvesting fertile material is not easily possible as large areas of the targeted frond can hold fruiting bodies. Confidence that recreational harvesters will have access to information on reproductive season was low.

Cutting height: Limit the proportion of the frond taken	Kelps	High – repeated throughout guidance, and supported by primary literature	Leaving the holdfast, stipe, and basal portion of the frond (meristem) is widely recommended and it is expected that the majority of harvesters follow this practice.
Cutting height: Limit the proportion of the frond taken	Browns	High – repeated throughout guidance and supported by primary literature	Leaving the holdfast, stipe, and basal portion of the frond is widely recommended and it is expected that the majority of harvested follow this practice. Extensive evidence for *A. nodosum*, although the recommended cutting height is variable.
Cutting height: Limit the proportion of the frond taken	Reds and Greens	Very high – repeated throughout guidance and supported by primary literature	Leaving the holdfast, stipe, and basal portion of the frond is widely recommended and it is expected that the majority of harvesters follow this practice.
Harvest areas sparsely, leaving unharvested plants between harvested individuals	Kelps	Medium - repeated throughout guidance and supported by some primary literature	Leaving unharvested individuals will minimise the ecological impact (e.g. wave dissipation, habitat provision), and will allow for more rapid recovery due to reproductive individuals remaining nearby. Confidence that recreational harvesters will have access to information on reproductive season was medium.
Harvest areas sparsely, leaving unharvested plants between harvested individuals	*A. nodosum*	Medium – consistent through most guidance and supported by primary literature	Leaving unharvested individuals will minimise the ecological impact (e.g. habitat provision). Confidence that recreational harvesters will have access to information was medium.

Harvest during the period of peak growth	Kelps	High – repeated throughout guidance and supported by primary literature	Confidence that recreational harvesters will have access to information on growing season is assessed as low.
Harvest during the period of peak growth	Reds and Greens	High – repeated throughout guidance and supported by primary literature	Harvesting during the season of peak growth allows for more rapid recovery of the canopy. Although confidence that most harvesters will have access to this understanding was medium, for most species (*Chondrus, Ulva* spp) this season is spring and summer, during which the majority of recreational harvesting interest if likely to be focussed.
Rotate harvested areas, leaving fallow years	*A. nodosum*	Very high – repeated throughout guidance and supported by extensive primary literature	Rotating harvested areas with fallow years between has been demonstrated to contribute to sustainability of the *A. nodosum* stock, although reported recovery times vary. It is expected that commercial harvesters will be aware of, and will apply, this guidance.

Appendix 18 Non native algae present in the UK

Seaweed INNS that may be proposed for harvesting and which case officers should be aware of. The presence of a species on this list does not suggest that the species is likely to be proposed for harvesting, have any commercial value or occur at sizes or abundances that make them suitable as target species. Evidence searches were conducted for all species for commercial harvesting and recorded.

Table 14. Seaweed INNS present in the UK that may be proposed for harvesting.

Scientific Name	**Phylum**	**Risk**	**Commercial exploitation**
Agarophyton vermiculophyllum	Rhodophyta	High	No evidence
Anotrichium furcellatum	Rhodophyta	No evidence	No evidence
Antithamnionella spirographidis	Rhodophyta	No evidence	No evidence
Antithamnionella ternifolia	Rhodophyta	No evidence	No evidence
Asparagopsis armata	Rhodophyta	High	Intentionally introduced to Europe as a food species. In Ireland: identified as a commercially important species for the production of cosmetics (Sweet 2011a).
Bonnemaisonia hamifera	Rhodophyta	Medium	No evidence
Botryocladia wrightii	Rhodophyta	No evidence	Of interest for cosmetics (Malakar and Mohanty, 2021
Caulacanthus okamurae	Rhodophyta	High	Grown Korea (Gao *et al.,* 2019)
Ceramium circinatum	Rhodophyta	No evidence	No evidence
Codium fragile subsp. Fragile	Chlorophyta	High	*C. fragile*-grown Korea (Hwang *et al.*, 2007), possible recreational harvesting UK

Scientific Name	**Phylum**	**Risk**	**Commercial exploitation**
Colpomenia peregrina	Ochrophyta	High	No evidence
Corynophlaea umbellata	Ochrophyta	No evidence	No evidence
Cryptonemia hibernica	Rhodophyta	No evidence	No evidence
Dasysiphonia japonica	Rhodophyta	High	No evidence
**Grateloupia subpectinata*	Rhodophyta	Medium	No evidence
Grateloupia turuturu	Rhodophyta	Medium	Yes- in parts of range
Melanothamnus harveyi	Rhodophyta	Medium	No evidence
Pikea californica	Rhodophyta	Medium	No evidence
Sarcodiotheca gaudichaudii	Rhodophyta	No evidence	Commercially important California (Pacheco-Ruíz, & Zertuche-González, 1996)
S. muticum	Ochrophyta	High	Aquaculture production in China (Liu *et al*., 2013) and a traditional food in Korea (Yang *et al*., 2013), there is no commercial use of this biomass for food consumption in Europe nor in rest of the non-native location (Lodeiro *et al*., 2004). Could be used as food, feed, and biofertilizers to agriculture crops (Milledge *et al*., 2016).
Solieria chordalis	Rhodophyta	Medium	Carrageenan yield, physicochemical properties, and antiviral activity of the carrageenan have been investigated (Boulho *et al*., 2017)

Scientific Name	**Phylum**	**Risk**	**Commercial exploitation**
Stenogramme interruptum	Rhodophyta	No evidence	No evidence
Ulva californica	Chlorophyta	No evidence	No evidence
Umbraulva dangeardii	Chlorophyta	No evidence	No evidence
U. pinnatifida	Ochrophyta	High	Yes-deliberately introduced to Brittany for commercial exploitation

Appendix 19 Invasive non native species supporting information

Table 15. INNS seaweeds that may be associated with harvested species and that are considered likely to cause significant ecosystem impacts: vector/pathway, distribution and key references.

Name	**Vector/pathway**	**Distribution (NBN atlas)**	**Key references**
Agarophyton vermiculophyllum	Spores and spermatia are non-motile and can only survive for up to a few days, so natural dispersal is limited. Probably: aquaculture shipments (Pacific oyster) to UK. Can occur as unattached and drifting mats. Further introductions may take place through transport of fragments via shipping or leisure craft. Secondary spread around GB is likely via dispersal of vegetative fragments by currents, via fishing and leisure craft, entanglement in fishing gear, migrating birds, or shellfish movements. In Sweden, spread was very rapid (150km in 2 years) (for references see Wood, 2019a)	Restricted: West coast only	Wood 2019a
Asparagopsis armata	The gametophyte stage attaches itself by its hooks to other material including fragments of other seaweeds and is dispersed through drifting or rafting on surface currents. The Falkenbergia phase spreads easily as floating balls. The two stages are thought to be spreading independently by vegetative means (clonal reproduction). Possible: aquaculture shipments (oysters) and shipping: Hull fouling.	Widespread: West and south coasts	Sweet 2011a
Bonnemaisonia hamifera	Possible: aquaculture shipments (oysters). Microscopic reproductive propagules are easily transported by water currents. Entire detached drifting plants and fragments of plants which are reproductively viable are also dispersed by currents whilst they remain buoyant, and may also become entangled in flotsam or fishing nets/anchor gear.	Widespread: West and south coasts	Sweet 2011b
Caulacanthus okamurae	Possible: aquaculture shipments (oysters). Likely: Shipping: hull fouling and ballast water vectors for spread to GB and around Europe. Floating fragments can be carried long distances. In GB most plants appear to be sterile so it is likely that reproduction is primarily vegetative by fragmentation and reattachment.	Restricted: West and south coast	Wood 2019b

Name	**Vector/pathway**	**Distribution (NBN atlas)**	**Key references**
Codium fragile subsp. fragile	Natural: produces a variety of propagules including vegetative buds, fragments of the thallus and entire dislodged thalli, all of which are dispersed by surface currents. It releases weakly swimming 'swarmer' cells toward the end of the growing season. Aquaculture: oysters. Shipping: fouling.	Restricted: mostly south coast	CABI
Colpomenia peregrina	Introduced to France aquaculture shipments of oysters. Natural spread from France to Britain has since occurred. Thalli are hollow, becoming air-filled and buoyant; this enables them to spread by floating on surface currents. Gametes released by the plant are dispersed in the water column before settling.	*Widespread*	Sweet, 2011c
Dasysiphonia japonica	Fragments can survive for long periods in ballast water. Secondary spread throughout Europe, probably the result of multiple vectors. It is thought to be spreading naturally in GB, primarily by the active shedding of small branchlets, which are transported in water currents, and by attachment to drifting algae. In addition, plastic debris, and recreational vessels are likely to be important anthropogenic vectors.	*Widespread: west coast*	Wood, 2021c
Grateloupia subpectinata	Likely: aquaculture shipments (oysters), Shipping: hull fouling. Spores may also be transported within ships' ballast water.	Restricted: South coast	Sweet 2011d.
Grateloupia turuturu	Likely: aquaculture shipments (oysters), Shipping: has been recorded attached to vessel's hulls, Spores may also be carried within ships' ballast water. Blades regenerate from old, damaged blades.	Widespread: South coast	Sweet, 2019a
Melanothamnus harveyi	Likely: transported with oysters.	Widespread: mainly west and south coasts	Eno *et al.,* 1997
Pikea californica	Possibly transported to the Isles of Scilly from California by flying boats during World War II.	Restricted: SW and Isles of Scilly	Sweet, 2019b
S. muticum	Aquaculture: oyster. Long range dispersal as drifting fragments become fertile while suspended. Germlings settle rapidly, usually near the parent. Detached fronds can continue to shed germlings as they drift.	Widespread	Sewell, 2019b

Name	**Vector/pathway**	**Distribution (NBN atlas)**	**Key references**
Solieria chordalis	No evidence	Restricted: mainly south coast	
U. pinnatifida	Deliberately introduced into the North Atlantic, to Brittany for commercial exploitation. Subsequent natural spread by propagules and drifting sporophytes attached to debris.	Widespread	Sewell, 2019a

Table 16 Attached or fouling INNS species that may be associated with harvested seaweed and are considered likely to cause significant ecosystem impacts: vector/pathway, distribution and key references.

Name	Vector/pathway	Distribution (NBN atlas)	Key references
Asterocarpa hu milis	Likely: aquaculture shipments (bivalves) Shipping: fouling species. The potential for natural dispersal is apparently very limited. Occasional natural rafting of adults on weed etc. is possible.	Restricted: West and south coast	Tillin *et al.,* 2020 (impacts) Bishop J. 2017.
Botrylloides diegensis	Natural dispersal ability probably very limited. Possible: dispersal by rafting of colonies on floating seaweed etc. Likely: aquaculture shipments (bivalves) Shipping: fouling species.	Restricted south coast	Bishop, 2011a
Botrylloides violaceus	Natural dispersal ability appears very limited: larvae with brief free-swimming period (4-10 h). Occasional dispersal by rafting of colonies on floating seaweed etc. is possible, and occurrence on *Cancer irroratus* suggests scope for dispersal by natural movement or commercial shipment of crabs. Likely to have been transferred long-distance with commercial oyster shipments, but also commonly fouls hulls and spread could occur by this mechanism.	Widespread	Bishop, 2012
Bugula neritina	Likely: Shipping: Introduction by hull fouling. Possible: Aquaculture-transport commercial oysters. Natural dispersal limited. Adult phase sessile and larva non-feeding (hence relatively brief motile phase).	Widespread	Bishop, 2011b
Ciona robusta	Limited natural dispersal. Probable: ballast tanks, fouling, Aquaculture: shellfish	Restricted: West and south coast	Yunnie & Bishop 2017
Cordylophora caspia	Larval dispersal is limited; Asexual reproduction by budding or fragmentation may be an important	Widespread	Sweet, 2019c

Name	**Vector/pathway**	**Distribution (NBN atlas)**	**Key references**
	factor in local dispersal. Rafting on floating debris may enhance long range dispersal. Possible introduced on foreign timber, Shipping; hull fouling and ballast tanks.		
Corella eumyota	Natural dispersal very limited: Probable Aquaculture (bivalves); Shipping: hull fouling and rapid spread along European Atlantic coasts, probably aided by leisure craft. Occasional rafting of adults on weed etc. possible.	Widespread	Bishop, 2019a
Diadumene lineata	Potential for natural dispersal in GB is apparently very limited due to the absence of sexual reproduction. The adult grows attached to a solid surface, and is thus potentially liable to rafting on natural or anthropogenic floating substrates.	Widespread: south and west coasts	Wood, 2020
Didemnum *vexillum*	Natural dispersal ability appears limited. Likely: aquaculture shipments (oysters), or Shipping: fouling Occasional dispersal by rafting with detached substrate possible. Outgrowths on floating structures can spread onto seabed below. Colony fragments can reattach and regrow, might be moved some distance.	Restricted: West and south coast	Bishop 2010: Impacts Tillin *et al.* (2020)
Hydroides ezoensis	Shipping: hull fouling	Restricted: South coast	Thorp *et al.* (1987)
Schizoporella japonica	Limited dispersal by swimming larvae. Possible: dispersal by natural rafting on weed. Aquaculture via stock transfers (oysters). Shipping: hull fouling and ballast water.	Restricted West and south	Wood, 2017
Styela clava	Limited natural dispersal Highly likely to be spread via aquaculture stock transfers and hull fouling.	Widespread	Bishop, 2019b

Name	**Vector/pathway**	**Distribution (NBN atlas)**	**Key references**
Tricellaria inopinata	Limited natural dispersal: Likely: shipping- hull fouling, Possible: Aquaculture -commercial oysters.	Widespread	Bishop, 2019b
Watersipora subatra	Limited natural dispersal. Possible: rafting on debris and seaweeds (has been found on drift *H. elongata*), Detached colonies survive and disperse living fragments and larvae. Shipping: hull fouling.	Restricted south coast	Bishop & Wood, 2021; impacts: Tillin *et al.,* 2020

Table 17. Mobile INNS species that may shelter in or be associated with harvested seaweed and are considered likely to cause significant ecosystem impacts: vector/pathway, distribution and key references.

Name	**Vector/pathway**	**Distribution (NBN atlas)**	**Key references**
Ammothea hilgendorfi	Natural dispersal is limited. Shipping: as the sea spider settles on developed fouling communities.	Restricted: South and east coast	Sweet, 2011e
Boccardia proboscidea	No information.	Restricted: South coast	
Caprella mutica	Aquaculture and shipping activities and recreational boating (while attached to hull fouling) as well as ballast water (Cook 2019 CABI). Field experiments confirmed the ability of C. mutica to disperse by rafting on drifting algae, with localised dispersal (< 5 km) by free-swimming (Ashton, 2006).	Widespread	Tillin *et al*. 2020 (impacts); Cook 2019-CABI datasheet; Ashton, 2006.
Cephalothrix simula	Possibly arrived in GB by natural planktonic drift from N. Europe, or by anthropogenic transport; Probably introduced to Europe via shipping or aquaculture (oysters)	No records	Wood, 2021
Hemigrapsus sanguineus	Natural larval dispersal by water currents .Probably introduced to European waters by Shipping-ballast water. Probable regional transportation: Shipping- hull fouling and aquaculture: oysters	Restricted: Southern	Sweet and Sewell, 2016
Hemigrapsus takanoi	Natural dispersal: larvae are planktonic (free swimming in the water column) for up to one month. Shipping: Larvae transported in ballast water, Possible: shipping- hull fouling; Aquaculture: transportation of oysters	Restricted: South east coast	Sweet, 2016
Palaemon macrodactylus	Recorded in the UK, the oldest European record is from the Thames in 1992.	No records on NBN	Ashelby, 2011

Urosalpinx cinerea	Limited natural dispersal- *U. cinerea* does not have a free swimming larval phase, Aquaculture: transport of oysters	Restricted: South and east coast only.	Tillin *et al*. 2020

Table 18. Summary of dispersal pathways for INNS seaweeds that may be associated with harvested species and whether seaweed harvesting has the potential to lead to increased dispersal (relevance).

Name	**Propagules**	**Drift**	**Attached to species**	**Attached to objects**	**Fragment regeneration**	**Mobile**	**Relevance**
Agarophyton vermiculophyllum	Limited	Yes	Yes	Yes	Yes	No	High
Asparagopsis armata	DD	Yes	Yes	Yes	Yes	No	High
Bonnemaisonia hamifera	Yes	Yes	Likely	Yes	Yes	No	Medium
Caulacanthus okamurae	No	Yes	DD	Yes	Yes	No	High
Codium fragile subsp. fragile	Possible	Yes	DD	Yes	Yes	No	Medium
Colpomenia peregrina	Yes	Yes	Yes	DD	DD	No	Medium
Dasysiphonia japonica	Yes	Yes	Yes	Yes	Yes	No	Medium
Grateloupia subpectinata	Likely	DD	Yes	Yes	Likely	No	High
Grateloupia turuturu	Likely	Likely	Yes	Yes	Likely	No	High
Melanothamnus harveyi	DD	Possible	Possible	Yes	DD	No	High
Pikea californica	DD	DD	DD	Yes	DD	No	High
S. muticum	Limited	Yes	Likely	Yes	Yes	No	High
Solieria chordalis	DD	DD	DD	DD	DD	No	High
U. pinnatifida	Yes	Yes	DD	Yes	No	No	Low

Table 19. Summary of dispersal pathways for attached and fouling INNS that may be associated with harvested species and whether seaweed harvesting has the potential to lead to increased dispersal (relevance).

Name	**Propagules**	**Drift**	**Attached to species**	**Attached to objects**	**Fragment regeneration**	**Mobile**	**Relevance**
Asterocarpa humilis	No	No	Yes	Yes	No	No	Medium
Botrylloides diegensis	Limited	DD	Yes	Yes	DD	No	Medium
Botrylloides violaceus	Limited	DD	Yes	Yes	DD	No	Medium
Bugula neritina	Limited	DD	Yes	Yes	DD	No	Medium
Ciona robusta	Limited	DD	Yes	Yes	No	No	Medium
Cordylophora caspia	Limited	Possible	Yes	Yes	Yes	No	High
Corella eumyota	Limited	DD	Yes	Yes	No	No	Medium
Diadumene lineata	No	DD	Yes	Yes	DD	No	Medium
Didemnum vexillum	Limited	Yes	Yes	Yes	Yes	No	High
Hydroides ezoensis	Yes	DD	Possible	Yes	No	No	Low
Schizoporella japonica	Limited	DD	Yes	Yes	DD	No	Medium
Styela clava	Limited	No	Yes	Yes	No	No	Medium
Tricellaria inopinata	Limited	No	Yes	Yes	No	No	Medium
Watersipora subatra	Limited	Yes	Yes	Yes	Yes	No	High

Table 20. Summary of dispersal pathways for mobile INNS species that may shelter in or be associated with harvested seaweed and whether seaweed harvesting has the potential to lead to increased dispersal (relevance). DD= Data deficient

Name	**Propagules**	**Drift**	**Attached species**	**Attached objects**	**Fragment regeneration**	**Mobile**	**Relevance**
Ammothea hilgendorfi	Limited	No	Yes	Yes	No	Limited	Medium
Boccardia proboscidea	Yes	DD	Yes	DD	DD	Limited	Low
Caprella mutica	No	Possible	Yes	Yes	No	Limited	Medium
Cephalothrix simula	No	No	Yes	Yes	No	Limited	Medium
Hemigrapsus sanguineus	Yes	No	Yes	Yes	No	Yes	Low
Hemigrapsus takanoi	Yes	No	Yes	Yes	No	Yes	Low

Name	**Propagules**	**Drift**	**Attached species**	**Attached objects**	**Fragment regeneration**	**Mobile**	**Relevance**
Palaemon macrodactylus	DD	DD	Possible	Possible	No	Yes	Low
Urosalpinx cinerea	No	No	Yes	Yes	No	Limited	Medium

Appendix 20 Invasive non native species: ecological impacts assessed using EICAT Methodology

Table 21. Assessment of ecological impacts of INNS seaweeds using the EICAT Methodology.

Name	Impact Assessment	Score	Reference
Agarophyton vermiculophyllum	Competition. Once established, can attain large biomass and these large populations could displace native macroalgae species such as fucoids. It is possible that it hinders native settlement by preventing substratum availability, and may cause mortality in larval stages by reducing oxygen and light availability (Maggs & Magill, 2014 and references therein; GISD, 2015; Hammann *et al*., 2013). However, in its invasive range in England and Ireland there is little evidence that it has had any negative effects. It tends to establish in muddy areas where there are few other algal species and it has not reached large enough biomass levels to adversely affect oxygen levels or water current movements.	Moderate	Tillin *et al*. 2020
Asparagopsis armata	Competition, Physical and Structural changes: Reported to dominate algal assemblages in some locations; it forms bloom-like outbreaks and is known to cover 100% of the upper infralittoral (0 – 10 metres depth) during winter in the NW Mediterranean	Major	Sweet, 2011a
Bonnemaisonia hamifera	Competition: While B. hamifera could potentially compete with other algae and seagrasses (Gollasch, 2009), in experiments B. hamifera showed a relatively slow growth rate and did not alter community biomass production rates (Sagerman *et al*., 2014). There is very little evidence in the literature of instances of competition with other algae No	Moderate	Sweet, 2011b

	evidence was found for impacts on aquaculture operations (see Sweet, 2011b for references).		
Caulacanthus okamurae	Physical and structural changes : can create a novel turf habitat in the upper intertidal zone where turfs did not previously exist; In studies in California, it displaced macroinvertebrates, such as limpets, periwinkles, and barnacles, but supported increased numbers of copepods and ostracods, and of fleshy seaweeds, including *Ulva* spp., *Gelidium*, and *Chondracanthus* (Smith *et al*., 2014). In Kent, within the Thanet SAC, *C. okamurae* has carpeted areas of the chalk reefs, a designated feature of the SAC (Tittley, 2014).	Moderate	Wood, 2019b
Codium fragile subsp. fragile	Competition: In some areas has altered community structure and composition. In the UK algal diversity is high and this species has not yet occurred in nuisance densities.	Minor	Sweet, 2011f
Colpomenia peregrina	Biofouling and Structural Impacts: May have the potential to smother species or cover areas of the shore, but no significant impacts have been reported on native species.	Minimal concern	Sweet 2011c
Dasysiphonia japonica	Competition: Moore and Harries (2009) described *D. japonica's* dominance at some sites in Scotland as a 'virtual monoculture'. In Norway, along the southwest coast it is now the most common species in sheltered and semi-exposed subtidal locations overgrowing other benthos. In addition, at some localities along the western Atlantic coast of N. America, can occupy up to 80 % of available space.	Massive	Wood, 2021c
Grateloupia subpectinata	Competition: No ecosystem impacts have been reported; however this large, fast-growing	Moderate	Sweet, 2011d

	seaweed may have the potential to displace native seaweed species.		
Grateloupia turuturu	Competition and physical impacts: No ecosystem impacts have been reported in Great Britain; however may have the potential to displace native seaweed species and shade neighbouring species. In North America Devil's tongue weed is a major competitor of *C. crispus* which provides an important winter food source for snails and other invertebrates. Winter die-back of Devil's tongue weed may therefore affect local ecology.	Moderate	Sweet, 2019
Melanothamnus harveyi	It possibly displaces native species as it can become very abundant, despite its small size	Moderate	Eno *et al*. 1997
P. californica	None known. Possible displacement of native species, but likely to be insignificant.	Minimal concern	Sweet, 2019b, Eno *et al*., 1997
S. chordalis	Competition. *Solieria chordalis* has been observed in the Gulf of Morbihan (France) since 2005 and in the Sarzeau peninsula (Morbihan, France) where strandings have become extremely abundant between July and October. During the summer over 15,000 t per year of red algae, mainly composed of S. chordalis, are removed. It is considered likely that this is impacting native seaweeds although no evidence was found.	Moderate	Boulho *et al*. 2017
S. muticum	Competition: out-competes native species because it is fast growing and reproduces within the first year of life by self fertilisation producing large numbers of offspring. Abundance has been correlated with reduction in diversity of native seaweeds and other species such as sea oak. Physical changes: Dense stands reduce light, increase sedimentation and alter temperature in rockpools.	Major	Sewell, 2019b

Undaria pinnatifida	Competition: likely to compete for space and resources with native species of kelp and other brown seaweeds. It may also compete with other epibenthic animals and seaweeds.	Major	Sewell, 2019a

Table 22. Assessment of ecological impacts of INNS attached and fouling species that may be associated with harvested seaweeds using the EICAT Methodology.

Attached/ fouling species	**Impact assessment**	**Score**	**References**
Asterocarpa humilis	Competition: A. humilis attaches to bivalves (Page *et al*., 2016) and associated substrates and is a possible competitor for food and space resources (Bishop, 2017). A. humilis could negatively affect other shallow-water suspension feeding sessile organisms. It may compete for resources and could impact on native species abundance (Bishop, 2017). Little is known at present about any impacts, like local species extinctions, that it may cause (Bishop, 2017).	Minimal concern	Tillin *et al*. 2020 (impacts) Bishop J. 2017.
Botrylloides diegensis	Competition: Biofouling: Capable of forming large colonies, and likely to have considerable effect on pre-existing sessile communities through overgrowth interactions etc. (Bishop (2011)	Major	Bishop, 2011a
Botrylloides violaceus	Competition: Biofouling: Capable of forming very large colonies, and likely to have considerable effect on pre-existing sessile communities through overgrowth interactions etc. Might therefore have a negative effect on the abundance and habitat occupancy of other shallow-water suspension feeding sessile invertebrates.	Major	Bishop, 2012
Bugula neritina	Competition: Biofouling: Populations in harbours and marinas can become dense, and colonies grow to considerable size. The species will thus presumably affect the abundance and habitat occupancy of other shallow-water suspension feeding sessile invertebrates. However, it is not clear whether this would cause the local extinction of any species.	Moderate	Bishop, 2011b
Ciona robusta	Potential Competition: co-occurs with the native Ciona intestinalis and possibly competes with it; additionally, limited natural hybridisation between C.	Minor	Yunnie & Bishop 2017

	intestinalis and C. robusta has been suggested in this region.		
Cordylophora caspia	Competition and structural changes: May compete with native species for space and food. This hydroid can form large, dense colonies which essentially modify the habitat and may affect benthic community composition.	Moderate	Sweet, 2019c
Corella eumyota	Competition: Corella eumyota populations might have a negative effect on the abundance and habitat occupancy of other shallow-water suspension feeding sessile invertebrates. However, it is not clear whether this would cause the local extinction of any species.	Minor	Bishop, 2019a
Diadumene lineata	Competition: D. lineata has been recorded in large clonal aggregations that could out-compete some native species (Podbielski *et al*. 2016), aggregations of over 4,000 individuals per square metre have been observed along from the Atlantic coast in the USA (Shick & Lamb, 1977). There is no evidence of any competitive impact from this species so this pathway has been assessed as of 'Minimal concern'	Minimal concern	Tillin and others 2020.
Didemnum vexillum	Competition, Biofouling and Physical and Structural changes: Competition: Competes with other sessile organisms for space and food whilst at the same time preventing epibenthic larvae from settling on it by lowering its surface pH. It smothers sessile communities and has a tendency to monopolize resources like space and food through its ability to rapidly colonise areas. Impacts are assessed as major where sessile organisms and algae may be overgrown and smothered and where competition is therefore focussed on space occupation.	Major	Bishop 2010: Impacts Tillin *et al.,* 2020
Hydroides ezoensis	Competition for space. This species is a severe fouling organism on harbour structures and ships' hulls throughout Southampton Water, adding considerably to fouling of poorly protected ships and causing buoyancy problems to buoys. Likely to compete with native fouling communities.	Moderate	Eno *et al*. 1997

Schizoporella japonica	Competition: This species is a competitor for space and is known to inhibit the growth of adjacent species. However it is a poor invader of previously occupied space.	Minor	Wood, 2017
Styela clava	Competition: can reach high densities, sometimes being the dominant species in shallow sheltered habitats. The species might thus have a negative effect on the abundance and habitat occupancy of other shallow-water suspension feeding sessile invertebrates. However, it is not clear whether this would cause the local extinction of any species. The relatively small holdfast takes up little space, while the tunic covering the body is often heavily incrusted by other sessile species.	Moderate	Bishop 2019b
Tricellaria inopinata	Competition, Biofouling: Populations in harbours and marinas can become very dense, with almost all submerged surfaces bearing a pale brown ‘fuzz’ of T. inopinata. Will thus presumably affect the abundance and habitat occupancy of other shallow-water suspension feeding sessile invertebrates. However, it is not clear whether this would cause the local extinction of any species. Also, kelps can become heavily fouled, particularly in sheltered sites, presumably increasing drag.	Moderate	Bishop 2019b
Watersipora subatra	Competition and Structural changes: This species can form large colonies overgrowing other sessile and encrusting species. This behaviour has the ability to alter the environment structure. It has been documented dominating fouling communities increasing habitat complexity with its growth forms and ability to retain sediments. This habitat alteration can have positive effects on species richness and diversity by providing structurally complex refugia. There is little evidence to suggest the ability of *W. subatra* to modify habitat structure has any negative impacts.	Minimal concern	Bishop & Wood, 2021; impacts: Tillin *et al.,* 2020

Table 23. Assessment of ecological impacts of mobile INNS species that may shelter in or be associated with harvested seaweeds using the EICAT Methodology.

Sheltering mobile species	**Impact assessment**	**Score**	**Key reference**
Ammothea hilgendorfi	No known impacts	Minimal concern	Sweet, 2011e
Boccardia proboscidea	Occasionally, worms have been reported associated with coralline algae. No evidence was found for impacts.	Data deficient	Simon *et al*. 2010 (habitat)
Caprella mutica	Biofouling: It has not been recorded fouling natural substratum around the UK. Competition: May compete with native caprellids, assessed as Major for those species but with impacts on ecosystem as lower.	Minor	Tillin *et al*. 2020 (impacts); Cook 2019-CABI datasheet; Ashton, 2006.
Cephalothrix simula	Predation: can affect food chains in invaded areas through predation on native species, and may also competitively exclude other predatory worms (Fernandez-Alvarez & Machordom, 2013). This species is one of the most common nemerteans along the coasts of Qingdao, China, and may be very abundant in certain intertidal habitats.	Moderate	Wood, 2021
Hemigrapsus sanguineus	Predation. Significant reductions in common shore crab abundance and mussel density have been reported where the Asian shore crab has achieved high densities in mainland Europe, and similar effects across the broader community may be expected. Common shore crab is also reported to have been displaced by the Asian shore crab in rocky shore habitats in several places in North America, including New England and New Jersey. Recruits and juveniles of other invertebrates including snails, barnacles and polychaetes may also be threatened due to increased predation	Moderate	Sweet & Sewell, 2016
Hemigrapsus takanoi	On a Dutch shore where the brush clawed shore crab has reached high densities, a drastic reduction in the number of juvenile native common shore crabs has been observed. Similarly in Dunkirk harbour this	Moderate	Sweet, 2016

	species appears to have replaced the common shore crab as the dominant and most abundant shore crab species. A similar impact on native crabs may occur were the brush-clawed shore crab to become established in GB.		
Palaemon macrodactylus	In San Francisco Bay it is thought to be out-competing native *Crangon* species but evidence for its impact on native species in other regions is lacking.	Data deficient	Ashelby, 2011- CABI Datasheet
Urosalpinx cinerea	Predation: *U. cinerea* is an active predator. Recorded prey species include at least 20 species of bivalves (including oysters and mussels), gastropods, barnacles, bryozoans, and small decapod crustaceans identified with prey preferences varying between populations. Predation impacts were considered to be moderate for habitats characterized by barnacles, *Mytilus edulis* and *Ostrea edulis* as many adults would be expected to have reached a size where predation is limited. However, at high densities and over longer time periods *U. cinerea* may reduce recruitment and impacts could be more severe.	Moderate	Tillin *et al*. 2020 (Impacts)

Appendix 21 Invasive non native species prioritised for management

Invasive non-native species were prioritised for management concern based on ecological impact, current distribution in the UK and the relevance of seaweed harvesting to spread, see section 3.4 for methodology.

Table 24. Invasive non-native species, prioritisation for management based on impact, distribution and relevance of seaweed harvesting to spread. Priority scores range from 1-5.

Species name	Impact	Distribution	Relevance	Priority
Agarophyton vermiculophyllum	Moderate	Restricted: West coast only	High	1
Asparagopsis armata	Major	Widespread: West and south coasts	High	2
Bonnemaisonia hamifera	Moderate	Widespread: West and south coasts	Medium	4
Caulacanthus okamurae	Moderate	Restricted: West and south coast	High	1
Codium fragile subsp. fragile	Minor	Restricted: mostly south coast	Medium	5
Colpomenia peregrina	Minimal concern	Widespread	Medium	5
Dasysiphonia japonica	Massive	Restricted: Widespread on west coast only	Medium	3
Grateloupia subpectinata	Moderate	Restricted: South coast	High	1
Grateloupia turuturu	Moderate	Widespread: South coast	High	2
Melanothamnus harveyi	Moderate	Widespread: mainly west and south coasts	High	2
Pikea californica	Minimal concern	Restricted: SW and Isles of Scilly	High	5
S. muticum	Major	Widespread	High	2
Solieria chordalis	Moderate	Restricted: mainly south coast	High	1
U. pinnatifida	Major	Widespread	Low	2
Asterocarpa humilis	Minimal concern	Restricted: West and south coast	Medium	4
Botrylloides diegensis	Major	Restricted south coast	Medium	3
Botrylloides violaceus	Major	Widespread	Medium	3
Bugula neritina	Moderate	Widespread	Medium	4
Ciona robusta	Minor	Restricted: West and south coast	Medium	5

Species name	**Impact**	**Distribution**	**Relevance**	**Priority**
Cordylophora caspia	Moderate	Widespread	High	2
Corella eumyota	Minor	Widespread	Medium	5
Diadumene lineata	Minimal concern	Widespread: south and west coasts	Medium	5
Didemnum vexillum	Major	Restricted: West and south coast	High	1
Hydroides ezoensis	Moderate	Restricted: South coast	Low	5
Schizoporella japonica	Minor	Restricted West and south	Medium	5
Styela clava	Moderate	Widespread	Medium	4
Tricellaria inopinata	Moderate	Widespread	Medium	4
Watersipora subatra	Minimal concern	Restricted south coast	High	5
Ammothea hilgendorfi	Minimal concern	Restricted: South and east coast	Medium	5
Boccardia proboscidea	Data deficient	Restricted: South coast	Low	5
Caprella mutica	Minor	Widespread	Medium	5
Cephalothrix simula	Moderate	No records (NBN)	Medium	1
Hemigrapsus sanguineus	Moderate	Restricted: Southern	Low	5
Hemigrapsus takanoi	Moderate	Restricted: South east coast	Low	5
Palaemon macrodactylus	Data deficient	No records	Low	Data deficient
Urosalpinx cinerea	Moderate	Restricted: South and east coast only.	Medium	3

Appendix 22 Case Study: Coastal-scale estimation of seaweed biomass for Wales

The total biomass of intertidal macroalgae in each region depends on the extent and area of habitat that potentially supports such growth. For Wales, for example, intertidal habitat extent is available as GIS data in the form of Ordnance Survey (OS) OpenData shapefiles showing the regions between low and high tide marks. This information can be combined with that of earlier mapping efforts by the Countryside Commission for Wales (CCW). CCW classed shore areas into substratum types also available as a GIS shapefile (Fig. 15 and 16). Combining these OS and CCW datasets allows the calculation of the area of intertidal foreshore that is rock, and thereby available for growth of seaweeds (Table 25).

Figure 15 Habitats by substratum type for Wales and Bristol Channel,

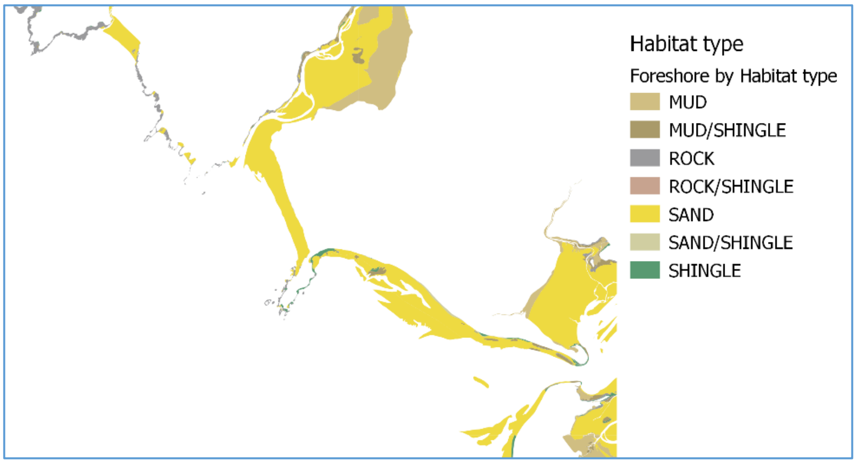

Figure 16 Habitats by substratum type for southwest Anglesey.

In other studies, and potentially applicable to this situation, habitat suitability can be predicted for each species using from regression models that link survey data to environmental predictors. Large-scale regional and national scale estimates of subtidal seaweed resources around Scotland have been made, for example, by scaling estimated habitat suitability to biomass using modelling methods as outlined in Burrows *et al*. (2018). While a complex modelling effort beyond the scope of this report, we can show how combining local scale biomass density data with environmental data can capture and refine estimates of the total biomass for each species at national level. As these estimates are based on an average area of shore in Scotland surveyed with quadrats in Scotland this model could be improved by collecting data from Welsh sites.

As part of a wider programme to examine the relationships between intertidal biodiversity, environment and climate, SAMS intertidal surveys between 2003-2010 recorded wet weights of all macroalgae species in 0.25m^2 quadrats placed 0.5m above MTL (“Upper”) and 0.5m below MTL (“Lower”) at 58 sites around Scotland and Northern England. Averages of these density estimates can provide a first approximation of the quantities available for harvesting. However, biomass density varies with the environment, with wave exposure influencing presence of species and abundance at local scales, water clarity and light attenuation (evidenced by remotely sensed ocean colour data) influencing at regional scales, and water temperature having effects at larger scales particularly for those species near the edge of their geographical distributions (Burrows *et al*., 2008; Vye *et al*., 2020). The combined effects of these environmental drivers on UK intertidal macroalgae can be seen in the resulting patterns of species richness from the concurrent surveys of presence and categorical abundance under the MarClim programme (Mieszkowska *et al*., 2005, Burrows *et al*., 2014). Numbers of macroalgae species recorded as present decreased with satellite estimates of chlorophyll a, and increased with wave exposure and average annual sea surface temperature.

Wave fetch values modelled at 200m scale (Burrows, 2012), chlorophyll a concentrations estimated from remote sensing (4.km-resolution MODIS Aqua satellite estimates from the NASA Giovanni data portal, averaged 2003-2018) and average annual surface temperature (SST: from OISST HR V2 0.5-degree data averaged 1982-2018) were collated to give the environmental context for the SAMS biomass density estimates. Each dataset was set into integer classes and converted from raster to vector formats, with the three resulting layers combined to give a single environmental shapefile (Fig. 16).

Locations of the SAMS biomass estimates were used to extract values for wave fetch, chlorophyll a and SST at each survey site. Mean biomass densities were evaluated for each class level for the three environmental variables.

Combining the environmental data layer with the intertidal habitat type layer allows the area of habitat to be associated with estimates of biomass density that are more specific to the environmental conditions. By comparing with species biomass totals derived from overall average densities and total habitat extent, we show how including environment-specific biomass estimates with the area of available rock in each of these conditions can strongly refine estimates of total biomass. The difference between these estimates is especially marked for those species with particular habitat requirements such as wave-shelter.

Biomass estimates using total rock habitat extent and average biomass densities

Combining the CCW habitat type and OS foreshore extent datasets gives the estimated area of habitat available for intertidal seaweeds in Wales as 36.6 km^2 (Table 25).

Table 25. Area of intertidal habitat in Wales

Habitat type	**Area (km^2)**
Sand/ shingle	7.73
Mud/ shingle	13.74
Shingle	16.85
Sand	336.62
Rock	36.57
Mud	132.6
Rock/shingle	0.49

Combining OS foreshore and Defra foreshore type shapefile layers allows estimation of the habitat available for intertidal seaweeds for England and Scotland. The total area across the two nations at 293 km^2, with potentially additional areas of 93 km^2 boulders and 25 km^2 rock with gravel (Table6). Most of the intertidal rock habitat is in Scotland (296/361 km^2 = 82%).

Table 26. Area of intertidal habitat in England and Scotland.

Habitat	**Area (km^2) England**	**Area (km^2) Scotland**	**Total area (km^2)**
Boulders/Loose Rock	4.9	44.3	49.2
Gravel	23.0	65.1	88.1
Made Ground (Man Made)	2.6	1.0	3.6
Mud	230.2	61.4	291.5
Mud & Gravel	11.8	13.6	25.3
Not Present	1.1	1.6	2.7
Rock Platform	61.8	231.2	293.1
Rock Platform With Banks Of Gravel	0.9	23.7	24.6
Rock Platform With Boulders/Loose Rock	2.7	41.5	44.2
Sand	235.0	274.5	509.5
Sand & Gravel	25.2	47.7	72.9
Sand & Mud	77.8	40.2	118.0
Unspecified	0.1	0.7	0.8

Average biomass per unit area was combined with area of appropriate substratum to give the potential total biomass of each species (Table 27). These biomass estimates use the simple approach of multiplying observed biomass per unit area by area of available substratum, without factoring in habitat suitability in terms of environmental conditions, such as wave exposure.

Table 27 gives wet biomass estimates of intertidal species using average values of wet weight per m^2 multiplied by total area of intertidal rock in Wales (the area represented above). For example, an average P. palmata biomass (wet weight) per unit area (per m^2) of 0.002 kg/m^2 multiplied by the total intertidal rock area in Wales in Table 26 gives an estimated total of 72336 kg of P. palmata, or 72.3 tonnes. For A. nodosum, the average biomass per unit area of 0.869 kg/m^2 yields a much higher estimate of 31,800 tonnes. Care must be taken in interpreting these estimates, as they take no account of the environmental conditions needed by each species. We address the effects of including these species-specific environmental requirements in the following section (Biomass estimates using habitat extent and environment-specific biomass values).

Table 27. Average biomass density estimates for seaweed biomass in Wales based on a rock habitat extent of 36.6km^2, using Scottish Association for Marine Science intertidal quadrat data (n=1750 per species, data collected 2003-2010)

Species	**Average kg/m^2**	**Lower kg/m^2**	**Upper kg/m^2**	**Estimated Biomass in Wales (tonnes)**
F. serratus	0.6690	1.3190	0.0190	24463
F. vesiculosus	0.9787	0.9799	0.9774	35788
A. nodosum	0.8694	0.9388	0.8001	31792
Porphyra spp	0.0588	0.1154	0.0023	2152
H. elongata	0.0405	0.0783	0.0028	1482
Enteromorpha spp	0.0387	0.0769	0.0004	1414
Laminaria spp	0.0055	0.0109	0.0001	201
M. stellatus	0.0064	0.0106	0.0021	233
F. spiralis	0.0419	0.0096	0.0742	1532
Corallina spp	0.0048	0.0095	0.0000	175
Polysiphonia spp	0.0016	0.0029	0.0003	58
P. palmata	0.0020	0.0014	0.0025	72
Cladophora spp	0.0002	0.0003	0.0002	9
C. crispus	0.0001	0.0002	0.0000	3

Biomass estimates using habitat extent and environment-specific biomass values

Environment-specific biomass densities combined with environment-specific rock extent improves estimates of total biomass. Using wave fetch as an example, Table 27 gives average biomass density across different environmental classes (from data collected in Scotland) and combines these biomass density estimates with extent information of wave exposure classes in Wales. The majority of Wales's rocky intertidal foreshore is moderately wave exposed (19.9 km^2 in wave fetch class 3.8). This means much less suitable habitat for wave-sheltered species such as the egg wrack *A. nodosum* and more for those species preferring wave-exposed shores, such as dulse *P. palmata*. This is reflected in the difference between estimates of the total biomass of species between those using species average biomass densities and environment-specific density estimates. Average biomass density gives a Wales-wide estimated total of 30000t wet weight of *A. nodosum* but accounting for environment-specific variation reduced this total to under 20000t. The reverse difference was seen for *P. palmata* with the environment specific values increasing the total estimate from 77 to 105t. Small differences between Table 26 and Table 27 result from the lack of environmental data for some sites in the SAMS survey dataset and across the Wales and SW England region of Fig. 15.

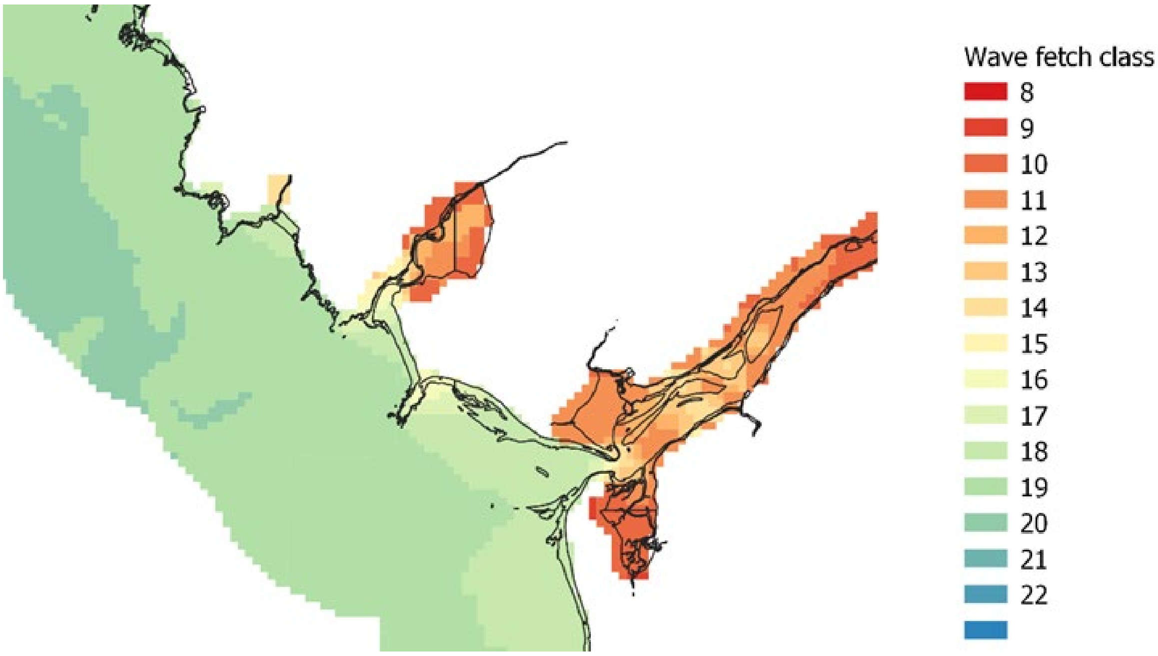

Figure 17. Wave fetch classes in southwest Anglesey.

Table 28 shows average biomass density of seaweeds from SAMS intertidal surveys, split by wave fetch classes from wave sheltered to very wave exposed. Area of rock habitat for Wales in each wave exposure category can be combined with wave-exposure specific estimates of biomass density to give the area-weighted and habitat-specific estimates of total biomass stocks (far right column).

Table 28. Average biomass of seaweeds (kg/m^2) from SAMS intertidal surveys in Scotland (2003-2010), split by wave fetch classes from wave sheltered to very wave exposed. NB. Total area of rock habitat differs slightly from Table 25 because of lack of environmental data for small stretches of coastline.

Species	**1.6**	**2**	**2.4**	**2.8**	**3.2**	**3.6**	**4**	**Average kg/m^2**	**Biomass in Wales: Using average kg/m^2**	**Biomass in Wales: Using habitat values**
F. vesiculosus	0.23	0.8791	0.7672	0.9157	1.2605	0.9735	1.3641	1.03	34999	34635
A. nodosum	9.92	2.4786	1.3740	0.8041	0.4922	0.2785	0.0495	0.87	29736	18920
F. serratus	0.094	0.8004	0.5052	0.5149	0.3375	0.6344	1.2685	0.68	23095	20003
Porphyra spp	0	0	0	0	0.1781	0.1863	0	0.07	2356	4777
H. elongata	0	0	0	0.0297	0.1572	0.0349	0.05	0.046	1563	1856
Enteromorpha	0	0	0	0.1176	0.0000	0.0930	0	0.045	1530	2435
F. spiralis	0	0.0061	0.1609	0.0653	0.0272	0.0102	0.0139	0.044	1483	778
Mastocarpus	0	0.0015	0.0068	0.0150	0.0062	0.0086	0.0028	0.007	252	291
Laminaria	0	0.0034	0.0000	0.0008	0.0000	0.0008	0.0283	0.006	212	63
Corallina officinalis	0	0	0	0	0	0.0233	0	0.006	189	463
Palmaria	0	0	0	0.0003	0.0141	0.0010	0	0.002	77	105
Polysiphonia	0	0.0015	0	0.0044	0.0021	0.0012	0	0.002	52	59
Cladophora	0	0	0.0006	0	0	0	0	0	3	0
Chondrus	0.0013	0	0.0003	0	0	0	0	0	2	1
Area of Rock in Wales, km^2	0.35	0.97	0.39	4.95	6.00	19.91	1.41	33.98	Not applicable	Not applicable
N quadrats	32	112	204	296	213	364	304	Not applicable	Not applicable	Not applicable

Appendix 23 Case Study - Cornish Seaweed Co. Sustainable commercial hand harvesting

The Cornish Seaweed Co. (CSC) have a licence from the Crown Estate and permission from intertidal landowners to harvest from a section of the south Cornish coast. They collect most species from the intertidal using scissors: *P. palmata*, *Ulva* spp., *S. latissima*, *L. digitata*, *L. hyperborea*, *Porphyra* spp., *C. crispus*, brown seaweeds (mostly *F. serratus* with some *F. vesiculosus*). *H. elongata* is also harvested by free diving.

Volumes are recorded by species as condition of licence, and the business maintains accurate records volumes and harvesting effort to monitor exploitation levels and resilience of seaweed stocks. CSC evaluate their own records on an annual basis as a guideline from which to monitor for declines which may precede overexploitation.

For all species harvested, the Natural England Guidance is followed (Bailey & Owen 2014). This generally involves harvesting sparsely and taking only part of the frond, leaving the base to regenerate (see section 4.1 on management approaches for further detail). Species are harvested seasonally during their peak in growth.

Areas of shore have been mapped into zones and harvesting is rotated between zones to allow for regrowth; no species is ever cleared from a large area at any time. Most sites are only visited once per season for each species, with the exception of *P. palmata* and *Ulva* spp. which appear to recover quickly, allowing for sustainable harvesting of most sites twice a year. At certain sites a proportion of the area is set aside as a no-take zone to serve as a spore donor population, promoting recovery.

To inform their own sustainability strategy, the standing stock biomass (of *L. digitata*, *H. elongata* and *P. palmata*) has been estimated at certain harvesting sites by conducting density and biomass calculations from quadrats, and mapping the extent of seaweed beds.

Biosecurity policy follows “check, clean, dry” protocol - all kit is washed and dried daily and only one site is visited and harvested per day.

CSC consider monitoring, both of the seaweed resource and associated ecosystem functioning, to be of great importance and would like to see it more widely supported. However, as it is often time consuming and expensive, feel it is unlikely to be widely performed on a voluntary basis.

Appendix 24 Seaweed lifecycle diagram

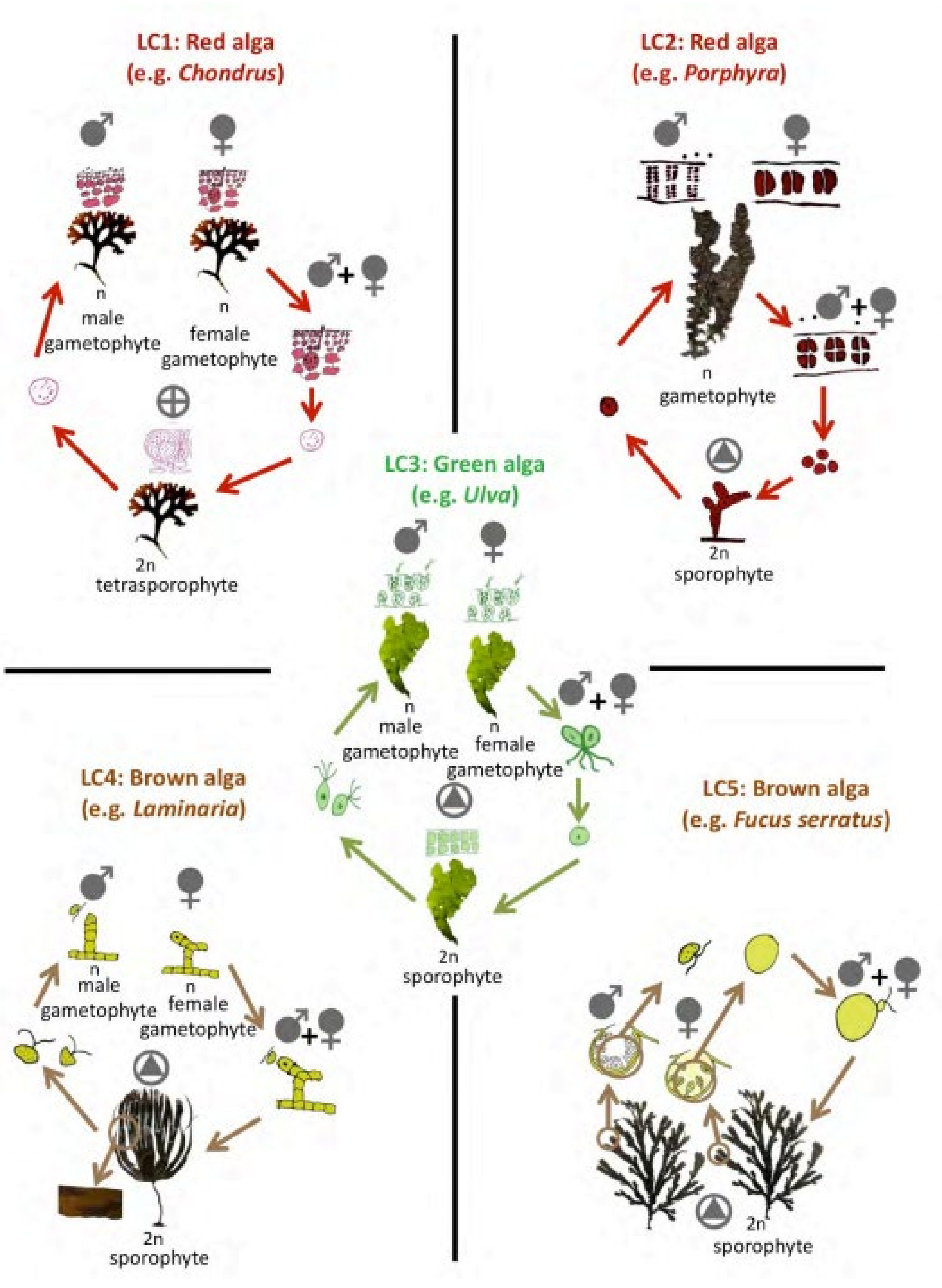

Figure 18. Seaweed reproductive strategies. Created by Jazmin Hernandez-Katun. Reproduced with permission from Edwards *et al*., 2012.

Data Archive Appendix

No data outputs were produced as part of this project.

Further copies of this report are available from library@cyfoethnaturiolcymru.gov.uk