Restoring marine and coastal habitats in Wales: identifying spatial opportunities and benefits

Report No: 554

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Report series: NRW Evidence Report

Report number: 554

Publication date: May 2021

Contract number: 554

Contractor: ABPmer

Contract Manager: Martin, A.

Title: **Restoring marine and coastal habitats in Wales: identifying spatial opportunities and benefits**

Author(s): Armstrong, S., Pearson, Z., Williamson, D., Frost, N., Scott, C.

Technical Editor: Frost, N.

Peer Reviewer(s) Bloomfield, H., Crook, D., Garrett, H., Hatton-Ellis, M., Lewis, H., Lindenbaum, K., Lough, T., Rimington, N., Vye, S., Wray, B.

Approved By: Frost, N.; Jackson, C. (ABPmer)

Lewis, M. (Natural Resources Wales)

Restrictions: None

Distribution List (core)

NRW Library, Bangor 2

National Library of Wales 1

British Library 1

Welsh Government Library 1

Scottish Natural Heritage Library 1

Natural England Library (Electronic Only) 1

Recommended citation for this volume:

Armstrong, S., Pearson, Z., Williamson, D., Frost, N., Scott, C. 2021. Restoring marine and coastal habitats in Wales: identifying spatial opportunities and benefits. NRW Evidence Report No: 554, 96pp, NRW, Cardiff.

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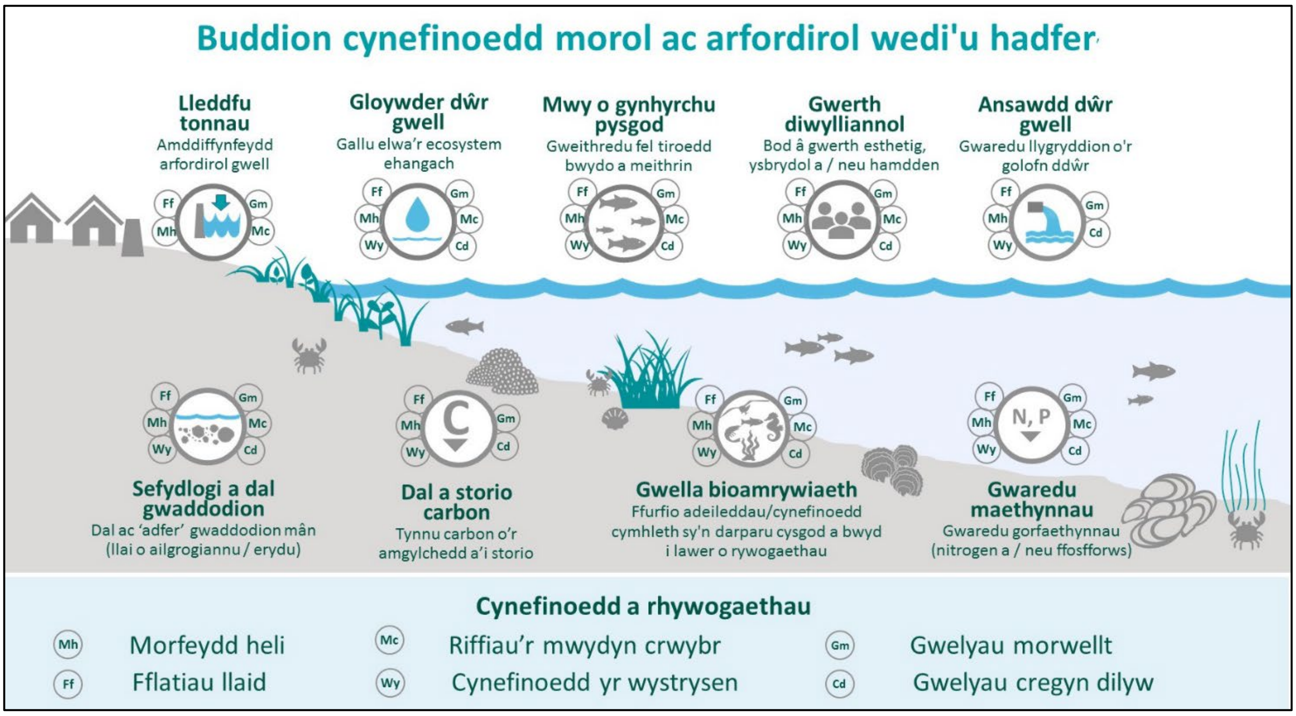
Crynodeb Gweithredol

Ar ran Cyfoeth Naturiol Cymru, ymgymerwyd â phrosiect ar y potensial sy'n bodoli i adfer cynefinoedd morol o amgylch Cymru, gan ganolbwyntio ar chwe chynefin gwerthfawr: fflatiau llaid rhynglanwol, morfeydd heli arfordirol, gwelyau morwellt, gwelyau cregyn dilyw, riffiau'r mwydyn crwybr a chynefin yr wystrysen frodorol. Prif ddiben y prosiect fu dod â gwybodaeth ynghyd, er mwyn llywio trafodaethau ehangach am gyfleoedd posibl ar gyfer gwaith adfer morol ac arfordirol yng Nghymru. Mae ffocws penodol wedi bod ar greu, a chasglu, mapiau neu gynhyrchion gofodol, gan mai ychydig o gynhyrchion gofodol/mapio yn bodoli sy'n helpu i nodi ardaloedd posibl lle gellir ymgymryd â gweithgareddau adfer a lle gellir eu canolbwyntio. Fodd bynnag, dylid pwysleisio y byddai angen i unrhyw brosiectau adfer ymgymryd â gwaith manwl i ddilysu'r safle a phroses o ymgysylltu'n lleol cyn i'r gwaith adfer gael ei wneud.

Gellir diffinio gwaith adfer fel gwaith sy'n cynnwys ailsefydlu prosesau naturiol, ymarferoldeb ecosystemau a bioamrywiaeth mewn cynefinoedd diraddiedig, yn ogystal ag ail-greu cynefin lle mae wedi cael ei golli cyn hyn.

Mae adolygiad llenyddiaeth cynhwysfawr wedi'i gynnal er mwyn darparu'r cyd-destun ar gyfer adfer morol, ei rôl mewn cynyddu cydnerthedd ecosystemau morol, a'r buddion lluosog y gall eu darparu. Canolbwyntiodd yr adolygiad hwn ar amrywiaeth o agweddau, gan gynnwys y ddeddfwriaeth ac ysgogwyr polisi allweddol, mecanweithiau/technegau ar gyfer cynnal gwaith adfer (gydag astudiaethau achos ledled Cymru, y DU a'r tu hwnt), a'r buddion a ddarperir gan gynefinoedd morol. Mae'r adolygiad wedi cadarnhau bod ffocws cynyddol ar adfer rhywogaethau a chynefinoedd morol, fel un dull gweithredu o wrthdroi'r dirywiad byd-eang mewn bioamrywiaeth. Gall adfer cynefinoedd coll a diraddiedig nid yn unig helpu i gyflawni dyletswyddau ac amcanion deddfwriaethol, ond gall hefyd arwain at nifer o fuddion i bobl Cymru a'r tu hwnt. Un ffordd arall o gyfeirio at fuddion gan gynefinoedd yw fel gwasanaethau ecosystemau. Mae'r chwe chynefin a astudir yma yn darparu gwasanaethau ecosystemau sylweddol a phwysig, gan gynnwys amddiffynfeydd arfordirol gwell, ansawdd dŵr gwell, a mwy o ddal a storio carbon, fel y dangosir yn y ddelwedd crynodeb isod.



Delwedd crynodeb o fuddion y gwasanaethau ecosystemau a geir gan gynefinoedd morol ac arfordirol sydd wedi'u hadfer

Mae'r holl gynefinoedd gwerthfawr hyn yn agored i niwed yn sgil effeithiau'r newid yn yr hinsawdd, yn bennaf trwy'r cynnydd cymharol yn lefel y môr, a hefyd trwy newidiadau i ynni gwynt a thonnau, tymheredd, a phatrymau glawiad. Byddai cynyddu neu gynnal eu maint trwy waith adfer yn mynd rhywfaint o'r ffordd i adeiladu cydnerthedd amgylchedd morol Cymru, er y cydnabyddir na all pob ymdrech adfer fod yn llwyddiannus yn y pen draw, neu barhau yn y tymor hwy.

Mae amrywiaeth o fylchau mewn gwybodaeth wedi'u hamlygu trwy gydol y prosiect hwn. Ar gyfer sawl cynefin, mae ansicrwydd sylweddol yn parhau ynghylch effeithiolrwydd tebygol mesurau creu/adfer cynefin morol posibl. Mae angen treialon, ymchwil a monitro cyson pellach i wella'r sylfaen dystiolaeth ac felly'r hyder sy'n bodoli wrth ymgymryd â gwaith adfer. At hynny, mae llawer o fylchau mewn gwybodaeth sy'n gysylltiedig â gwasanaethau ecosystemau yn bodoli, o ran cynefinoedd brodorol penodol (gyda llenyddiaeth yn gymharol brin ar welyau pysgod cregyn er enghraifft) a'r gwasanaethau eu hunain, yn ogystal â gwerthoedd ariannol cysylltiedig.

Mae sawl cynnyrch gofodol/mapio wedi'u cynhyrchu neu eu cyfuno yn rhan o'r prosiect hwn, a fydd yn ddefnyddiol iawn wrth amlygu lle mae cyfleoedd i adfer a chreu cynefinoedd morol yn bodoli yng Nghymru. Mae'n bwysig nodi, fodd bynnag, nad yw'r mapiau o reidrwydd yn nodi y bydd adfer yn ymarferol neu'n hyfyw yn ariannol mewn lleoliad penodol. Fodd bynnag, dylai'r mapiau ddarparu ffocws ar gyfer trafodaeth ac ymchwil pellach i'r potensial ar gyfer gwaith adfer mewn rhai o'r ardaloedd hyn.

Mae'r cynhyrchion mapio hyn yn cynnwys y canlynol:

- Haen sy'n nodi ardaloedd yn y gorlifdir a allai fod yn addas ar gyfer creu fflatiau llaid a morfa heli trwy adlinio a reolir (er nad yw categoreiddio ardaloedd fel hyn yn golygu y byddant o reidrwydd yn ymarferol / yn hyfyw yn ariannol);

- Haenau sy'n amlygu ardaloedd lle gall cyfleoedd / yr amodau cywir fodoli ar gyfer morwellt, cynefin yr wystrysen frodorol ac adfer gwelyau cregyn dilyw (mae dwy o'r haenau hyn wedi'u creu gan asiantaeth arall ar gyfer y DU gyfan; sylwer nad ystyriwyd bod creu haen ar gyfer *Sabellaria alveolata* yn angenrheidiol ar yr adeg hon).

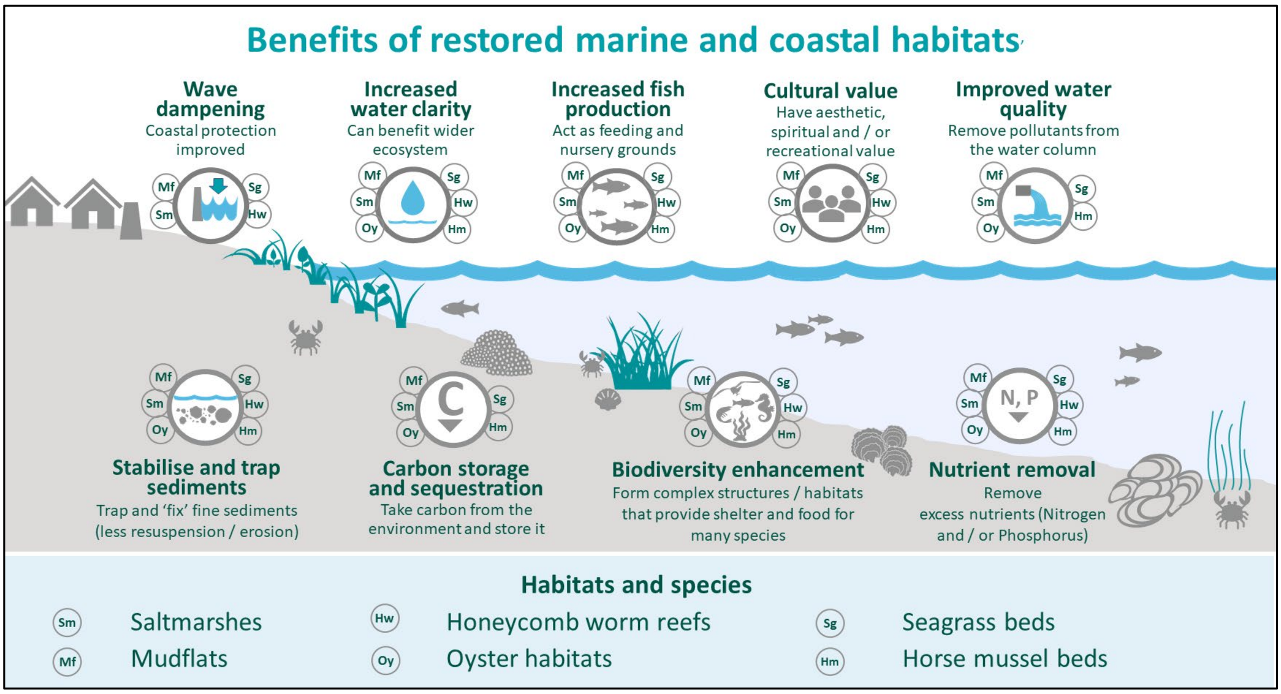
Dylid ystyried pob un o'r haenau data hyn yn gymhorthion cychwynnol i nodi lleoliadau neu ardaloedd posibl, ac maent yn dod â sawl cyfyngiad a amlygwyd yn yr adroddiad, y mae angen eu hystyried yn llawn wrth ddefnyddio'r cynhyrchion hyn. Gan yr ystyrir y prosiect hwn yn un o'r camau cyntaf wrth nodi ardaloedd a allai fod yn addas ar gyfer adfer cynefinoedd arfordirol a morol yng Nghymru, bydd angen cynnal astudiaethau manwl a gwaith ymgysylltu lleol bob amser cyn dechrau gwaith ar unrhyw leoliadau penodol i'w hadfer. Dylid hefyd ymgynghori â chanllawiau perthnasol, lle bônt ar gael, a dylid cyflogi arbenigwyr lle bo hynny'n briodol.

Executive summary

On behalf of Natural Resources Wales, a project on marine habitat restoration potential around Wales has been undertaken, focussing on six valuable habitats: intertidal mudflats, coastal saltmarshes, seagrass beds, horse mussel beds, honeycomb worm reefs and native oyster habitat. The main purpose of the project has been to bring together information, in order to inform wider discussions about potential opportunities for marine and coastal restoration in Wales. A particular focus has been to create, and gather, maps or spatial products, as few spatial / mapping products exist which help identify potential areas where restoration activities can be undertaken and focussed. However, it should be emphasised that any restoration projects would need to undergo detailed site validation and local engagement before restoration is carried out.

Restoration can be defined as including both the re-establishment of natural processes, ecosystem functionality and biodiversity in degraded habitats, as well as re-creating habitat where it has previously been lost.

A comprehensive literature review has been undertaken in order to provide the context for marine restoration and its role in increasing the resilience of marine ecosystems and the multiple benefits it can provide. This review focussed on a variety of aspects, including the key legislation and policy drivers, mechanisms / techniques for restoration (with case studies of projects across Wales, the UK and beyond), and benefits provided by marine habitats. The review has confirmed that, increasingly, there is a focus on restoration of marine species and habitats, as one approach to reversing the global decline in biodiversity. Restoring lost and degraded habitats can not only help meet legislative duties and objectives, but also harness multiple benefits for the people of Wales and beyond. Benefits from habitats may also be referred to as ecosystem services. The six habitats studied here provide substantial and important ecosystem services; including improved coastal protection, better water quality and more carbon storage and sequestration, as shown in the summary image below.



Summary image on ecosystem service benefits of restored marine and costal habitats

All of these valuable habitats are vulnerable to the impacts of climate change, mostly through relative sea level rise, and also changes to wind and wave energy, temperature and rainfall patterns. Increasing or maintaining their extent through restoration would go some way to building the resilience of the Welsh marine environment; though it is recognised that not all restoration efforts may ultimately be successful, or persist longer term.

Various knowledge gaps have been highlighted throughout this project. For several habitats, considerable uncertainties remain about the likely efficacy of possible marine habitat creation / restoration measures. Further trials, research and consistent monitoring are required to improve the evidence base and thus confidence in undertaking restoration. Furthermore many ecosystem service-related knowledge gaps exist, both with regard to specific native habitats (with literature relatively scarce on shellfish beds for example), and services themselves, as well as related monetary values.

Several spatial / mapping products have been produced or brought together as part of this project, which will be very useful in highlighting where opportunities for marine habitat restoration and creation exist in Wales. It is important to note, however, that the maps do not necessarily indicate that restoration will be feasible or financially viable in a given location . However, the maps should provide a focus for further discussion and investigation of the potential for restoration in some of these areas.

These mapping products include:

- A layer indicating areas in the floodplain that might be suitable for the creation of mudflat and saltmarsh through managed realignment (though areas being categorised as such does not mean they will necessarily be feasible / financially viable) ;
- Layers highlighting areas where opportunities / the right conditions may exist for seagrass, native oyster habitat and horse mussel bed restoration (two of these layers have been created by another agency for the whole of the UK; please note that the creation of a layer for *Sabellaria alveolata* was not considered necessary at this point).

All of these datalayers should be considered as initial aides to identifying potential locations or areas, and come with several limitations which have been highlighted in the report, and which need to be fully considered when using these products. As this project is considered to be one of the first steps in identifying areas that could be suitable for coastal and marine habitat restoration in Wales, detailed studies and local engagement will always need to be undertaken before pursuing any specific locations for restoration. Relevant guidebooks, where available, should also be consulted, and specialists employed where appropriate.

1. Introduction

The Well-being of Future Generations Act 2015 and the Environment (Wales) Act 2016 formalised Welsh Government's commitment to the sustainable management of natural resources (SMNR) and their objective to halt and reverse the decline in biodiversity, as well as maintaining and enhancing the resilience of ecosystems. Increasingly, there is a focus on the restoration of species and habitats, as one approach to reversing the global decline in biodiversity. For example, the United Nations (UN) Decade on Ecosystem Restoration, which started in 2021, is a rallying call for the protection and revival of ecosystems all around the world, for the benefit of people and nature. Restoring lost and degraded habitats can not only help meet legislative duties and objectives but also harness multiple benefits for the people of Wales and beyond.

Few spatial products exist which help identify areas where restoration activities can be undertaken and focussed. Thus, this project was commissioned to identify spatial opportunities for restoration of marine and coastal habitats and species, and also provide an assessment of the wider benefits of such restoration, in Welsh waters. This work also aimed to bring together existing evidence and inform a wider discussion in Wales about the potential opportunities and benefits of restoration at sea.

Restoration in the context of this project includes both the re-establishment of natural processes, ecosystem functionality and biodiversity in degraded habitats, as well as re-creating networks of habitat where these have been lost (International Union for Conservation of Nature (IUCN), 2019).

The key objective of this project was to, where appropriate, develop and/or signpost a series of datalayers to demonstrate the spatial opportunities for the restoration of marine and coastal habitats and species in Welsh waters. The focus has been on the following six habitats / species: intertidal mudflats, coastal saltmarshes, seagrass beds, horse mussel beds, *Sabellaria alveolata* reefs and native oyster habitats. These habitats are all of principal biodiversity importance in Wales. To provide the context for marine restoration and its role in increasing the resilience of marine ecosystems, a literature review has also been undertaken to document a variety of aspects, including the key legislation and drivers, mechanisms / techniques and ecosystem services.

This report is structured as follows.

- Section 1: Introduction;
- Section 2: Methodology;
- Section 3: Literature review (focussed on ecology and ecosystem services related to the habitats / species, as well as drivers and techniques for restoration);
- Section 4: The opportunity datalayers (and how they may be used); and
- Section 5: Conclusions and recommendations.

Maps depicting the opportunity datalayers can be found in Section 4, and maps for all the other created datalayers have been provided in Appendix A.

2. Methodology

2.1 Introduction

As noted above, this project involved the following key tasks:

(1) The mapping of opportunities in relation to six marine habitats/species; and

(2) A literature review.

The methods employed to undertake these tasks are now briefly outlined in turn in Sections 2.2 and 2.3.

2.2 Opportunity mapping

The key objective of this project has been to identify and, where appropriate / not already available, develop a series of datalayers which demonstrate the spatial opportunities for the restoration of six marine and coastal habitats and species in Welsh waters. These six habitats are: intertidal mudflats, coastal saltmarsh, seagrass beds, horse mussel beds, *Sabellaria alveolata* reefs and native oyster habitats.

The following iterative steps have been undertaken:

1. Compile / signpost data on existing and historic extent / mapping of the habitats / species in Wales. Create / compile historic datalayers where possible. This was informed by a workshop and discussions with experts from Natural Resources Wales (NRW) and academia.

2. Compile information / evidence on habitat requirements (drawing on the literature review, see next Section).

3. Compile information on similar / existing opportunity datalayers. This has served as a gap analysis.

4. Develop methodologies for datalayer creation. This was informed by a workshop and detailed discussions with NRW experts.

5. Datalayer creation. Following the gap analysis, this focussed on the creation of two datalayers, native oyster, and a combined mudflat / saltmarsh datalayer. Furthermore, two dedicated average tidal flow and wave datalayers have been created as input datalayers, by extracting relevant data from ABPmer's tide and wave hindcast models. Please note that a beneficial use-related datalayer has also been created; this shows sources of fine sediment for saltmarsh and mudflat recharge in Wales.

Tables 1 to 3 below summarise what historic, input and opportunity mapping / datalayers have been compiled and created for the purpose of this project. Tables 4 and 5 then provide high level information on the processing steps for the two dedicated datalayers which have been produced. Maps depicting the opportunity datalayers can be found in Section 4 (including two related layers created by the Joint Nature Conservation Committee (JNCC)). Maps for all of the other created layers are provided in Appendix A.

Table 1. Historic datalayers created as part of the project

Subject / Species	Layer created?	Detail
Native oyster habitat	Yes	One point layer each, derived from OBIS and NBN. Shows records of individuals, not nec. reefs. Age of record categorised.
Horse mussel habitat	Yes	Same as above
Sabellaria alveolata reefs	Yes	Same as above
Seagrass meadows	No	-
Mudflats	No	-
Saltmarshes	Yes	Combined polygon layer for selected estuaries (collating previous efforts only)

Table 2. Opportunity datalayers created as part of the project

Subject / Species	Layer created?	Detail
Native oyster habitat	Yes	One point layer each, derived from OBIS and NBN. Shows records of individuals, not nec. reefs. Age of record categorised.
Horse mussel habitat	No	UK wide layer being developed by JNCC
Sabellaria alveolata reefs	No	Not considered to be required
Seagrass meadows	No	UK wide layer being developed by JNCC
Mudflats & Saltmarshes	Yes	One combined polygon layer for both habitats, showing opportunities on current floodplain (for method overview, see next table)

Table 3. Input layers created for this project

Subject	Detail
Tidal Energy	One raster datalayer, showing (depth averaged) mean spring tidal flow speeds
Wave Energy	One raster datalayer, showing average annual significant wave height
Beneficial use	One point datalayer, showing potential soft material sources for beneficial use, including types of material (maintenance dredge disposal sites)

Table 4. Key processing parameters applied to creating native oyster opportunity datalayer

Factors / Predictors	Criteria	Input layer(s)
Bathymetry / depth to seabed	0 to 80 m	EMODnet bathymetry
Tidal energy / current speed	Less than 1 m s^{-1}	Dedicated layer produced for this project (see above)
Substrate	Mixed and coarse sediments, mud, hard silt, rocks, muddy sand.	JNCC marine habitat data product: EUNIS level 3 combined map

Table 5. Key processing parameters applied to creating mudflats and saltmarshes opportunity datalayer

Factors / Predictors	Criteria	Input layer(s)
Floodplain	`All tidal & tidal / fluvial areas selected. Foreshore, beaches and areas already subject to tidal influence removed.	Welsh / NRW Zone 3 Floodmap
SMP Policy	For all 3 epochs, (above) floodplain areas were overlapped with the different policies.	Welsh / NRW Shoreline Management Plan layer (available from Lle Geoportal)

2.3 Literature review

A comprehensive literature review has been undertaken, and is presented in Section 3; this documents the following:

- the key legislation and drivers with respect to habitat restoration;
- the ecology of the six habitats, and their underlying physical / chemical requirements;
- the main mechanisms for delivering restoration / creation of the habitats in practice; and

the wider benefits that could potentially be obtained by delivering restoration of these habitats and species.

3. Literature review

3.1 Introduction

This literature review firstly presents background key legislation and drivers for marine habitat restoration in Section 3.2, before Section 3.3. discusses the ecology and habitat requirements of the six habitats / species which are the focus of this report. Section 3.4 then reviews key mechanisms and techniques for their restoration, and related natural accounting and ecosystem services are outlined in Section 3.5 (with a summary table provided in Section 3.5.3).

3.2 Key Legislation and Drivers for Marine Habitat Restoration

3.2.1. International Drivers

There are a myriad of drivers for marine habitat restoration. Many international drivers exist, and have often informed / fed into Welsh legislation and policy, which is outlined further below. Notable recent international initiatives, which the UK and Wales are part of, include the following:

- The Leaders' Pledge for Nature: Political leaders participating in the United Nations Summit on Biodiversity in September 2020, representing 84 countries from all regions and the European Union, committed to reversing biodiversity loss by 2030.
- The Global Ocean Alliance and the '30 by 30' initiative: An initiative / alliance joined by over 30 countries (as of November 2020), which is pushing for at least 30% of the global ocean to be protected in Marine Protected Areas (MPAs) by 2030.
- The Post-2020 Global Biodiversity Framework: This builds on the Strategic Plan for Biodiversity 2011-2020 and sets out an ambitious plan to implement broad-based action to bring about a transformation in society's relationship with biodiversity and to ensure that, by 2050, the shared vision of living in harmony with nature is fulfilled.
- The 2021 United Nations Climate Change Conference, also known as COP26: The 26th United Nations Climate Change conference is scheduled to be held in Glasgow, Scotland, in November 2021 under the presidency of the UK.
- The United Nations (UN) Decade on Ecosystem Restoration (2021-2030): This was proclaimed by the UN General Assembly following a proposal for action by over 70 countries. Its aim is the building of a strong, broad-based global movement to ramp up restoration and put the world on track for a sustainable future. This includes political momentum for restoration, as well as thousands of initiatives on the ground.

It is also worth noting that the restoration of the six habitats which have been the focus of this project can be interpreted as constituting nature-based solutions (Image 1). The IUCN defines such solutions as '*actions to protect, sustainably manage, and restore natural and modified ecosystems that address societal challenges effectively and adaptively, simultaneously providing human well-being and biodiversity benefits*'.

University of Oxford, 2021

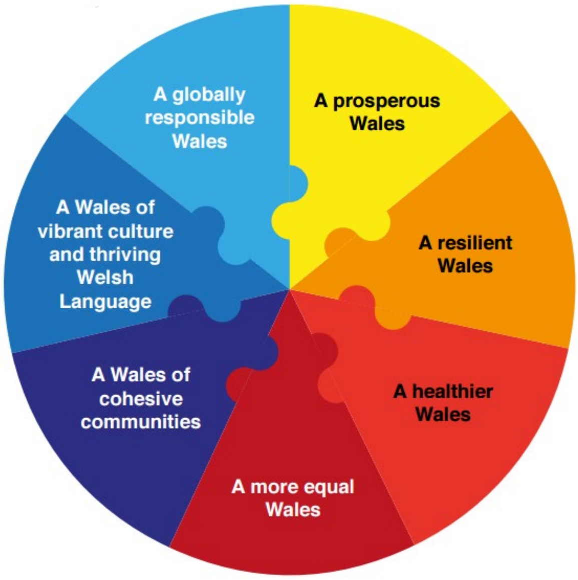
Image 1. Conceptual diagram of nature-based solutions

3.2.2. Welsh legislation, policy and plans

Key Welsh policies, legislation and plans with regard to marine habitat restoration are now outlined below, to provide the context for marine restoration and its role in increasing the resilience of marine ecosystems.

Well-being of Future Generations (Wales) Act 2015

The Well-being of Future Generations (Wales) Act 2015 seeks to improve the social, economic, environmental and cultural well-being of Wales. This Act put in place seven well-being goals which are outlined in Image 2. Under the 'Resilient Wales' goal, it aims to create '*a nation which maintains and enhances a biodiverse natural environment with healthy functioning ecosystems that support social, economic and ecological resilience and the capacity to adapt to change (for example climate change).*'



Welsh Government, 2015

Image 2. Seven wellbeing goals of the Well Being of Future Generations (Wales) Act

Environment (Wales) Act 2016

The Environment (Wales) Act 2016 sets out the requirement for the 'sustainable management of natural resources'. While 'restoration' is not explicitly outlined, Sections 3, 4, 6 and 7 of the Act are the key sections which are of relevance to 'restoration'. Section 3 on the 'sustainable management of natural resources' notes that objective is to;

'maintain and enhance the resilience of ecosystems and the benefits they provide and, in so doing—
(a) meet the needs of present generations of people without compromising the ability of future generations to meet their needs, and
(b) contribute to the achievement of the well-being goals in section 4 of the Well-being of Future Generations (Wales) Act 2015 (anaw 2).'

Section 4 sets out principles for the sustainable management of natural resources, and notes the "principles of sustainable management of natural resources" are to;

'(a)manage adaptively, by planning, monitoring, reviewing and, where appropriate, changing action;
(f)take account of the benefits and intrinsic value of natural resources and ecosystems;
(i)take account of the resilience of ecosystems, in particular the following aspects—
(iv)the condition of ecosystems (including their structure and functioning).'

Section 6 of the Act requires public authorities to seek to '*maintain and enhance biodiversity […] in the exercise of their functions'*. Section 7 additionally requires Welsh Ministers to publish a list of living organisms and habitats in Wales, which are considered of key significance to sustain and improve biodiversity in relation to Wales Section 7 of the Act.

The Act also notes that the Welsh Ministers must '*take all reasonable steps to maintain and enhance the living organisms and types of habitat included in any list published under this section and encourage others to take such steps'*.

All of the six habitats / species which are the focus of this report are listed as being 'of principal importance' under Section 7 of the Environment (Wales) Act 2016.

The 2017 Natural Resources Policy

The Natural Resources Policy focusses on improving the way Wales' natural resources are managed. This is a key part of the delivery framework for the sustainable management of natural resources, as established by the Environment (Wales) Act.

The policy highlights three main challenges faced with regard to natural resources as follows:

- Improving ecosystem resilience;
- Climate change and the decline in biological diversity; and
- The UK's withdrawal from the EU.

In addition, three national priorities for the management of Wales' natural resources are listed, namely:

- Delivering nature-based solutions;
- Increasing renewable energy and resource efficiency; and,
- Taking a place-based approach.

The 2020/21 Nature Recovery Action Plan for Wales

Welsh Government published the national biodiversity strategy 'The Nature Recovery Action Plan for Wales' in 2015, and refreshed it in 2020/21. Its ambition is to '*halt the decline in biodiversity by 2020 and then reverse the decline, for its intrinsic value, and to ensure lasting benefits to society*. The Plan sets out how Wales will deliver the commitments of the UN convention on biological diversity, the strategic plan for biodiversity 2011-2020 and the 20 associated Aichi targets (a short term framework for action), as well as the EU biodiversity strategy. It confirms that resilient ecological networks are vital for nature recovery.

The Plan focusses on six objectives for nature recovery in Wales, and actions to reverse the decline of biodiversity are set out under each objective. The objectives are as follows:

- Objective 1: Engage and support participation and understanding to embed biodiversity throughout decision making at all levels.
- Objective 2: Safeguard species and habitats of principal importance and improve their management.
- Objective 3: Increase the resilience of our natural environment by restoring degraded habitats and habitat creation.
- Objective 4: Tackle key pressures on species and habitats.
- Objective 5: Improve our evidence, understanding and monitoring.
- Objective 6: Put in place a framework of governance and support for delivery.

The actions are allocated to specific partners, including public bodies and local nature partnerships. Public bodies are required to consider using the Plan as a basis on which to base a 'biodiversity and ecosystem resilience duty forward plan'.

Planning Policy Wales (2016 / 2018)

Welsh terrestrial planning policy is outlined in the Planning Policy Wales (PPW), which was first published in 2016 (and last updated in 2021). The primary objective of PPW is '*to ensure that the planning system contributes towards the delivery of sustainable development and improves the social, economic, environmental and cultural wellbeing of Wales, as required by the Planning (Wales) Act 2015, the Well-being of Future Generations (Wales) Act 2015 and other key legislation*' (Welsh Government, 2018). PPW includes specific policies on conserving and enhancing the natural environment through planning. It states that the planning system should contribute to the delivery of sustainable development and improve the social, economic, environmental and cultural well-being of Wales. The PPW and the associated 2021 National Plan 'Future Wales' concentrate on development and land use issues of national significance, indicating areas of major opportunities and change, highlighting areas that need protecting and enhancing and helping to co-ordinate the delivery of Welsh Government.

Under Section 2 (Achieving Well-being Through Placemaking), key considerations, including those in relation to the environment, are set out. This notes that, as part of their decision-making processes planning authorities should consider whether the resilience of ecosystem will be improved. As part of Section 6 (Recognising the Special Characteristics of Places) of the planning policy, it is ruled that; '*to ensure mechanisms allow for the identification of potential habitat, the maintenance of existing assets and networks and promote the restoration of damaged, modified or potential habitat and the creation of new habitat'* must be implemented. Furthermore, the same paragraph of the PPW states that '*planning decisions should incorporate measures which seek the creation, restoration and appropriate management….'.* For connectivity to be implemented, one should seek to '*develop functional habitat and ecological networks within and between ecosystems and across landscapes, building on existing connectivity and quality and encouraging habitat creation, restoration and appropriate management. The opportunities could include enlarging habitat areas, developing buffers around designated sites…*'.

Technical advice notes (TANs) have been produced in connection with the PPW, for example, TAN 5 is on 'nature conservation and planning'.

Future Wales – the 2021 National Plan

Future Wales – the National Plan 2040 is the Welsh national development framework, setting the direction for development in Wales to 2040. Future Wales is a spatial plan, which means it sets a direction for where there should be investment in infrastructure and development for the greater good of Wales and its people. Future Wales' defines 11 'Outcomes', which are overarching ambitions to be achieved by 2040, based on the national planning principles and national sustainable placemaking outcomes set out in PPW.

Outcomes 9 to 11 are particularly pertinent for this report; these aim to achieve '*A Wales where people live*':

- (9) '... *in places that sustainably manage their natural resources and reduce pollution'*;
- (10) '... *in places with biodiverse, resilient and connected ecosystems'*; and
- (11) '... *in places which are decarbonised and climate-resilient*'.

The 2019 Welsh National Marine Plan (WNMP)

The Welsh National Marine Plan (WNMP) was prepared and adopted under the Marine and Coastal Access Act 2009 and in conformity with the UK Marine Policy Statement. It represents the start of a process of shaping Welsh seas to support economic, social, cultural and environmental objectives. Its overarching objective is to

> *'Support the sustainable development of the Welsh marine area by contributing across Wales' well-being goals, supporting the Sustainable Management of Natural Resources (SMNR) through decision making and by taking account of the cumulative effects of all uses of the marine environment'.*

Under the topic 'living within environmental limits', the following key objectives of note to this report are as follows:

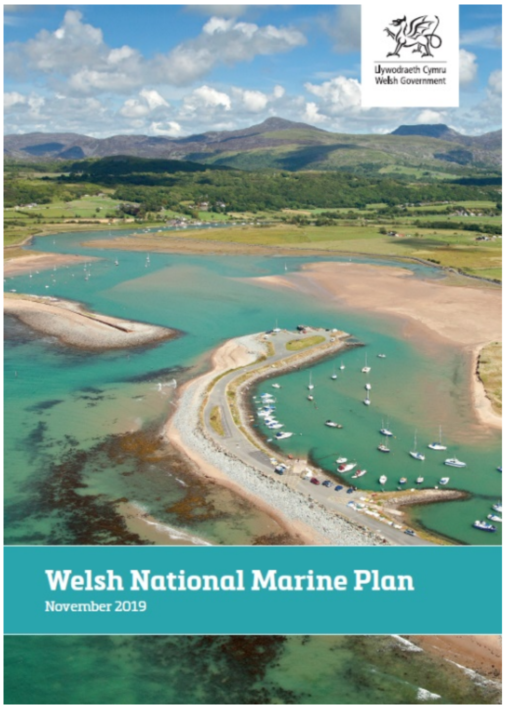


Image 3. WNMP Title page

- *'Support the achievement and maintenance of Good Environmental Status (GES) and Good Ecological Status (GeS).*
- *Protect, conserve, restore and enhance marine biodiversity to halt and reverse its decline including supporting the development and functioning of a well-managed and ecologically coherent network of Marine Protected Areas (MPAs) and resilient populations of representative, rare and vulnerable species.*
- *Maintain and enhance the resilience of marine ecosystems and the benefits they provide in order to meet the needs of present and future generations.'*

Many of the policies require the observation of the Environmental Impact Assessment (EIA) mitigation hierarchy. Notably, Policy ENV_01 on 'Resilient marine ecosystems' states:

> *'Proposals should demonstrate how potential impacts on marine ecosystems have been taken into consideration and should, in order of preference:*
>
> *a. avoid adverse impacts; and / or*
> *b. minimise impacts where they cannot be avoided; and / or*
> *c. mitigate impacts where they cannot be minimised.*
>
> *If significant adverse impacts cannot be avoided, minimised or mitigated, proposals must present a clear and convincing case for proceeding.*

'*Proposals that contribute to the protection, restoration and / or enhancement of marine ecosystems are encouraged.*'

Policy ENV_01 aims to ensure that biological and geological components of ecosystems are maintained, restored where needed and enhanced where possible, to increase the resilience of marine ecosystems and the benefits they provide.

Also particularly noteworthy is policy ENV_02 on MPAs, which states:

'Proposals should demonstrate how they:

> *a. avoid adverse impacts on individual Marine Protected Areas (MPAs) and the coherence of the network as a whole;*
> *b. have regard to the measures to manage MPAs; and*
> *c. avoid adverse impacts on designated sites that are not part of the MPA network.'*

There is also a table in the WNMP detailing the Plan policies that support the achievement of Good Environmental Status under the Marine Strategy Framework Directive, and the achievement of Water Framework Directive goals are referenced in connection with Policy ENV_06 on 'Air and water quality'.

In 2020, Welsh Government published an implementation guidance document to support the effective and consistent implementation of WNMP policies..

3.2.3. European Transposing Regulations and the Welsh MPA network

The regulations on EIA, WFD and MSFD referenced in the WNMP have been transposed into various pieces of UK and Welsh regulations / legislation, which are not listed separately here. However, these can all drive marine habitat creation to varying degrees, as can other legislation related to European transposing regulations, notably the Birds and Habitats Directives. The latter seek to establish a network of protected sites, which are known as Special Protection Areas (SPAs) and Special Areas of Conservation (SACs). Marine SACs and SPAs, as well as other internationally and nationally protected sites make up Wales' MPA network, which is what WNMP policy ENV_02 refers to. There are 139 MPAs[1] in Welsh waters, that are made up of:

- 13 Special Protection Areas (SPAs);
- 15 Special Areas of Conservation (SACs);
- 1 Marine Conservation Zone (MCZ);
- 107 Sites of Special Scientific Interest (SSSIs); and
- 3 Ramsar sites.

Many of these MPAs contain one or more of the six habitats which form the focus of this report as designated features. Where these are purposely damaged as part of a development, compensation may need to be provided (ABPmer, 2020). Where the MPA

[1] It should be noted that the number of sites within the MPA network is reported differently (140) by Welsh Government, 2018. This is assumed to be a function of the SSSI features that are considered to be coastal / marine within the respective counts. The latest details with respect to designations should be obtained from Lle – A Geo-Portal for Wales (inshore and coastal), or JNCC's Protected Area Datasets (offshore).

features are in unfavourable condition, then management action may be undertaken to restore or enhance a given habitat.

It is worth noting that, post-Brexit, the provisions of the above mentioned directives generally remain, as they are implemented in national law, with adjustments to account for Brexit. For example, the Habitats Regulations have been amended by The Conservation of Habitats and Species (Amendment) (EU Exit) Regulations 2019, which mirror existing provisions.

3.3 Habitat Background for The Six Habitats

For each of the six habitats which are the focus of this report, some background information is now provided, including Welsh context, status, ecology, and sensitivities and tolerances (including in relation to potential impacts arising from climate change).

3.3.1. Saltmarsh and mudflat

Welsh context

There are 76 km² of saltmarshes and 434 km² of intertidal sand and mudflats in Wales (NRW, 2020). Both habitats are widespread across the Welsh coast, where they are present in all major estuaries and inlets as well as in other more sheltered locations, including the lee of spits and in the shelter of islands (Welsh Government, 2018). The largest extents are found in the estuaries of Carmarthen Bay, whilst the Dee and Severn Estuaries also contains extensive stretches of saltmarsh.

Saltmarshes and mudflats are listed in Annex I of the Habitats Directive and are also both 'habitats of principal importance' under Section 7 of the Environment (Wales) Act 2016. These habitats also frequently form a major component of two encompassing 'habitat' features, namely 'estuaries' and 'large shallow inlets and bays'. Many Welsh mudflats and saltmarshes are furthermore protected as features of Ramsar sites, SACs or SSSIs, and / or constitute supporting habitats for the bird interest features of many SPAs. Intertidal mudflats are also listed on the OSPAR Convention list of 'threatened and/or declining species and habitats' in the North-East Atlantic.

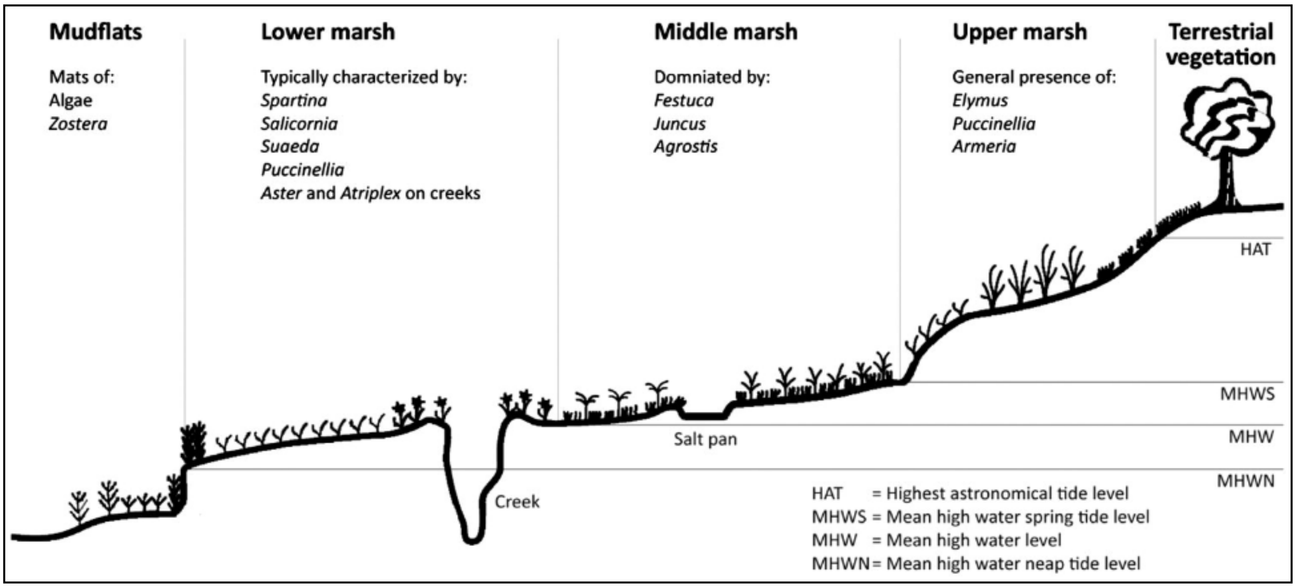
Historical trends

Nationally and globally, there have been extensive historic losses of these intertidal habitats, chiefly due to human activities, mostly related to land claim, and the construction of sea defences, ports and harbours. It has been estimated that approximately 100,000 ha of British saltmarshes were lost between 1600 and 1900, mainly to gain more land for agricultural production (Toft *et al.*, 1995). More recently, comparatively small-scale industrial and residential developments have been the main driver for land claim (Adam, 2002). Available historic mapping for saltmarshes shows that UK wide trends are replicated in Wales. For example, a comparison of saltmarsh extent at Rumney to Peterstone, east of Cardiff shows that, between the early 1950s and 2017, 22% of the saltmarsh had been lost along this Severn Estuary frontage (NRW, 2021a). This erosion will at least in part have been due to a process known as 'coastal squeeze', which is mostly related to sea level rise [2].

Ecology

Intertidal mudflats and saltmarshes are generally understood to develop between the levels of the highest astronomical tides (HAT) and mean low water springs (MLWS) on tidal, sheltered, coasts, where the predominant sediments are silts and muds. These fine sediments are kept in suspension where currents are fast, and settle when the currents are slowest (French, 1997).

Tidal flats are un-vegetated 'banks of mud or sand that are exposed at low tide', which generally slope gently seawards. Image 4 illustrates how, as elevation increases, and thus tidal inundation frequency decreases, vegetation can become established and saltmarshes develop throughout temperate regions (Trenhaile, 1997). Sediments needed for this elevation increase ('accretion') can be derived from marine, coastal, and fluvial sources, as well as *in situ* reworking (Pethick, 1984).



Source: Davis *et al.*, 2018

Image 4. Generalised division of intertidal habitats based on elevation in relation to tidal height

The more shelter is afforded to mudflats and saltmarshes, from both wave and tidal energy, the more accretion / sedimentation is likely to take place. Saltmarsh plants can play a crucial role in such shelter (e.g. Adam, 2002), as can higher topographical features, such as islands and bunds, which can provide protection from waves and wind, and/or slow tidal currents. Elevation and inundation frequency is also important, as it is only whilst the habitat is covered in water (the so-called 'hydroperiod') that sediment laden water can reach it.

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2 Coastal squeeze is '*the loss of natural habitats or deterioration of their quality arising from anthropogenic structures, or actions, preventing the landward transgression of those habitats that would otherwise naturally occur in response to sea level rise in conjunction with other coastal processes. Coastal squeeze affects habitat on the seaward side of existing structures*' (Defra et al., 2021).

The lower lying a site, the more frequently inundated it is, and the higher accretion rates tend to be. For example, French (2006) quotes the example of a Severn Estuary low saltmarsh which accreted at a rate of 12.1 mm yr^{-1}, whereas on the highest level marsh (which would only be covered by high spring tides and surges), the rate was much reduced at 2.3 mm yr^{-1}. Conversely, the higher a site is, the more likely plant roots will have established to hold sediments together and prevent re-suspension / erosion during high wind / wave conditions, or on occasions where heavy rainfall coincides with low tide. Furthermore, there needs to be sediment available for accretion. In some Welsh estuaries, water column suspended sediment concentrations (SSC) can be very low, and accretion across intertidal habitats thus minimal. Conversely, in some estuaries, notably the Severn and Dee Estuaries, SSC tend to be very high, such that accretion can be rapid in sheltered locations. Suspended sediment loads are not constant in any given estuary, they are known to vary depending on tidal state, tidal height, freshwater flows, season and weather (e.g. Uncles *et al.*, 2004). SSC are furthermore invariably higher at the bed than at the water surface (e.g. Kirby, 1986).

Saltmarshes are generally divided into several zones; these zones broadly correspond to the frequency of tidal inundation and the associated effects of salinity and tidal scouring. Four main saltmarsh zones can be distinguished, and are commonly referred to as ‘pioneer’, ‘low’, ‘middle’ and ‘upper’ or ‘higher’ marsh. These saltmarsh types are typically associated with a characteristic number of tidal inundations per year (Table 6), which can broadly be related to tidal levels as shown in Image 4.

Table 6. Inundation frequency associated with intertidal habitats

Inundations Per Year	**Habitat**
More than 450	Mudflat
450 to 360 (maximum continuous exposure: 9 days; minimum daily daylight submergence of 1-2 hours)	Pioneer marsh
Less than 360 (typical 300) (minimum continuous exposure: 10 days; maximum daily daylight submergence of 1 hour)	Low, mid and upper/higher saltmarsh

Source: Nottage and Robertson, 2005; Toft *et al.*, 1995

There are approximately 40 species of higher plants found in saltmarsh habitats, with any individual saltmarsh containing between 10 and 20 species, although in a broader sense there are a larger number of plant species found at the upper and transitional zones (Boorman, 2003). The National Vegetation Classification (NVC) system recognises 28 communities of saltmarsh vegetation, which include the following (NRW, 2019a):

- Pioneer communities (i.e. early colonisers, beginning a chain of ecological succession), which are dominated by species such as such as *Spartina anglica* (often present as a result of deliberate introduction historically) and *Salicornia* spp.;
- Lower and middle marsh communities, comprised of species such as *Puccinellia maritima, Atriplex portulacoides* and *Limonium vulgare*;
- Mid to upper marsh communities, dominated by species such as *Festuca rubra* and *Juncus maritimus*; and
- Upper marsh communities, including species such as *Agrostis stolonifera*, *J. maritimus* or *Elytrigia altherica*.

Additional plant communities which can also be present on saltmarsh are covered by the NVC system as well; these include certain inundation grassland types, brackish reedbeds and swamp communities and mires (Rodwell 1991, 1992 and 1995).

Each saltmarsh species has a different tolerance to tidal flooding, and therefore a different, although often overlapping, vertical range. *Spartina* is the saltmarsh species most tolerant of tidal inundation, and its distribution was hence found to be slightly less closely related to tidal levels than other plants (Clarke *et al.*, 1993).

Saltmarsh zonation can also be impacted by other variables; these, however, tend to be less important than the inundation frequency / hydroperiod discussed above. For example, the nature of the sediment may influence the elevation that saltmarsh species occur. On sandier substrates, lower marsh zones tend to be at higher elevations due to the lower nutrient contents of sand (Adam, 2002). In larger estuaries, and estuaries with a larger tidal range, the limits of species also tend to be farther up the shore than predicted by the level of MHWN alone. This is due to the generally greater degree of exposure to wind and wave action, increased velocity of flows and higher turbidity variation (Clarke *et al.*, 1993; Leggett *et al.*, 2004). Chemical factors can also have an influence; for example, excessively waterlogged sediments due to over-consolidation can lead to low oxygen diffusion rates, and consequently low sediment redox potential and retarded plant establishment (e.g. Garbutt *et al.*, 2006).

Sensitivity and tolerance

With regard to salinity tolerance, saltmarsh plants exhibit a variety of tolerances, with more sensitive species typically found in upper to high marsh zones. Where salinities are lower, further up estuaries or where there are transitions to freshwater communities, brackish reedbeds would typically develop. Mudflats can establish in very low and high salinity areas, with the invertebrate communities inhabiting them changing according to salinity regime.

Saltmarshes are not sensitive to changes in oxygenation, but are slightly sensitive to changes in nutrient levels / enrichment and heavy metal contamination. These habitats are furthermore moderately sensitive to contamination by synthetic compounds and are highly sensitive to hydrocarbon contamination (i.e. from oil spills) (MarLIN, 2001; 2004).

Mudflats and saltmarshes face various threats from climate change; the primary threats are relative sea level rise, changes to wind and wave energy, temperature and precipitation. Where sufficient sediment is present in a given system’s water column, then these habitats are relatively resilient and can keep pace with sea level rise to various extents by accreting / increasing in height. However, their landward movement is often impeded, and over time, they would be expected to be ‘squeezed’ as more and more of their seaward margins become subtidal. This retreat could be further exacerbated by storm damage, particularly if such events become more frequent, as storms often cause substantial intertidal habitat erosion, particularly along their seaward edges. Upper saltmarsh areas conversely frequently benefit from storm events as large volumes of sediment may be delivered. Changes in temperature could lead to the decline in cover for several invertebrate and marsh plant species, as suitable conditions shrink with rising temperatures (Marine Climate Change Impacts Partnership (MCCIP), 2018a).

3.3.2. Seagrass beds

Welsh context

There are around 7.3 km² of mapped seagrass beds in Wales, with the largest extents found in the Severn Estuary, as well as around Anglesey and in Milford Haven. Seagrasses are deemed as scarce in Wales (present only in 16–100 ten km squares) (Stewart and Williams, 2019), although not necessarily declining. NRW (2016) note that '*intertidal seagrass beds have increased in extent*' although the timescales over which this change has occurred are unclear.

Seagrass beds are a habitat of principal importance under Section 7 of the Environment (Wales) Act 2016. *Zostera* beds are also listed as threatened and declining under OSPAR. Many Welsh seagrass beds are located within designated sites, where they constitute a component part of larger features such as 'estuaries' or 'large shallow inlets and bays', although they are not an 'Annex I' habitat in their own right. Notable seagrass habitats are for example included in the Pembrokeshire Marine / Sir Benfro Forol and Severn Estuary SACs. In addition, seagrass is listed as a feature in nine SSSIs, including Milford Haven Waterway and the Traeth Lafan SSSI (Welsh Government, 2018).

Source: Andy Pearson

Image 5. Seagrass

Historical trends

Globally, seagrass meadows are declining at an unprecedented rate (Waycott *et al.*, 2009; Orth *et al.*, 2006). In the UK, a series of research papers have clearly defined seagrasses to be under threat and in a perilous state (Jones and Unsworth, 2016; Jones *et al.*, 2018; Unsworth *et al.*, 2017). Seagrass was thought to be once very abundant and widespread around the British coasts, but serious declines have occurred, in particular due to poor water quality (eutrophication and other pollutants), land claim and a severe outbreak of a suspected 'wasting disease' in the early 1930s (Davison and Hughes, 1998; Butcher, 1933; Green *et al.*, 2021). Such an outbreak of disease was probably exacerbated by poor estuarine and coastal water quality (Short and Wyllie-Echeverria, 1996). The extent of the damage in Wales is not well known, as seagrass was poorly mapped before the disease struck (Brown, 2015).

Recovery of eelgrass beds in the UK has been slow and patchy, with loss still continuing in many places. However, cases of recovery have occurred, such as substantial expansion within intertidal *Z. noltii* beds in Milford Haven in West Wales (Bertelli *et al.*, 2017), and slight expansion with substantial density increases in the Skomer MCZ (Burton *et al.*, 2019).

Ecology

Seagrass beds develop in intertidal and shallow subtidal areas, in areas sheltered from significant wave action. Two species of *Zostera* occur in the UK; eelgrass (*Z. marina;* often used interchangeably with *Z. angustifolia*[3]) and Dwarf eelgrass (*Z. noltii*). *Z. marina* is the largest of the British seagrasses (with trailing leaves up to 1 m long) and typically occurs in the shallow sublittoral down to 4 m depth, in fully marine conditions and on muddy to relatively coarse sediments (occasionally with a mixture of gravel) (Davison and Hughes, 1998; Dale *et al.*, 2007). Dwarf eelgrass, *Z. noltii* occurs higher on the shore, on the intertidal and upper extent of the subtidal, on substrates of fine, detritus-rich sand and mud.

Please note that widgeon grass (*Ruppia* spp.) is a genus of aquatic freshwater plants found in the UK including Wales, that have similar environmental preferences to *Zostera* spp., and are sometimes treated as seagrasses. The two species of widgeon grass found in the UK (beaked tasselweed, *R. maritima* and spiral tasselweed, *R. cirrhosa*) are not strictly considered as part of the traditional seagrass arrangement (Kuo and Den Hartog, 2001), but they are commonly grouped with *Zostera* spp.[4] as they can occupy a similar niche due to their pronounced salinity tolerance (Zieman, 1982)..

Zostera spp. are flowering plants (angiosperms) adapted to saline conditions. These plants have stems (rhizomes) that spread horizontally below the sediment surface, and shoots that grow above the surface forming expansive ‘meadows’ in both the intertidal and shallow subtidal zones. In contrast to other marine vegetation (such as macroalgae), seagrasses flower, develop fruit and produce seeds like terrestrial plants. They also have roots and a vascular system that transports gases and nutrients around the plant (NRW, 2019b).

Seagrass leaves of both *Zostera* species slow down water currents / flow rates under the canopy and encourage the settlement of fine sediments, detritus and larvae, which in turn stabilises the sediments and protects against wave disturbance. This increases the species diversity within the seagrass bed, with *Z. marina* beds in particular attracting wrasse and goby species (also associated with kelp beds), pipe fish, sea anemones, neogastropods, prosobranch molluscs and brooding cuttlefish.

Sensitivity and tolerance

In the UK, *Z. marina* is most commonly restricted to a maximum of 7 m water depth, however the maximum known depth of *Z. marina* in the British Isles is 10 m (Dale *et al.*, 2007). *Z. noltii* is an intertidal species. *Z. marina* prefers enclosed areas including embayments, lagoons and estuaries, whilst *Z. noltii* is commonly found in straits, sea

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3 There is some discussion around the taxonomy of this species.

4 Please note that *R. maritima* is included in the definition of Biodiversity Action Plan (and by extension Section 7) seagrass habitat (JNCC, 2008a).

lochs, estuaries, lagoons and embayments. Both *Z. marina* and *Z. noltii* can tolerate very weak (negligible), weak (<1 knot) or moderately strong (1-3 knots) currents. They are highly sensitive to increases in wave exposure, which may increase erosion and result in the uprooting and loss of individual eelgrass plants. An increase in current speed may be beneficial to eelgrass beds exposed to low currents and high suspended sediment concentrations however, as high suspended sediment can decrease light penetration and therefore decrease photosynthesis and growth. The shallow depth distribution of *Z. marina* is the result of its high light requirements as a photosynthetic organism. Estimates from across a range of *Z. marina* suggest it requires between 12 and 37% of surface irradiance (SI) to survive in the long-term with a mean %SI of 18 (Lee *et al.*, 2007 and Erftemeijer and Lewis, 2006). There is limited literature on the light requirements of *Z. noltii*, though what information is known indicates that Z. noltii is tolerant of both low and high light intensities (Lee *et al.*, 2007; Erftemeijer and Lewis, 2006) explaining the capacity of *Z. noltii* to live exposed at low tide and submerged in estuarine conditions that are commonly turbid.

Although *Z. marina* requires marine conditions, it is able to cope with reduced levels of salinity and has a variable salinity range of 18-40 psu. Evidence suggests common eelgrass struggles to cope if salinities are low for prolonged periods of time (Salo *et al.*, 2014) however, its preferential range is 30 to 40 psu. *Z. noltii* is more tolerant to large fluctuations in salinity, but has a narrower salinity tolerance range of 18-30 psu. *Z. noltii* is highly resistant to desiccation and so can be found high up in intertidal zones, but also permanently submerged in brackish or estuarine environments (Charpentier *et al.*, 2005). *Z. marina* are moderately sensitive to local increases in temperature beyond 25°C, at which point photosynthesis becomes inhibited and photosynthetic rates may decrease by as much as 50% (Nejrup and Pedersen, 2007; cited in D'Avack *et al.*, 2019). *Z. noltii* meanwhile is not sensitive to minor increases in temperature, and is tolerant of temperatures between 5 and 30°C. *Z. marina* and *Z. noltii* are moderately sensitive to nutrient and organic enrichment, as this inhibits the plants' ability to remove and recycle nutrients (MarLIN, 2005; 2019).

With regard to potential impacts from climate change, effects on sea temperature, sea level, storminess and rainfall patterns could affect seagrass species. These impacts could be both positive and negative, and could lead to effects on seagrass productivity, growth and flowering rates, as well as habitat distribution. For example, warmer winters may lead to less damage during exceptionally cold winters, whereas hotter summers could lead to more beds suffering from a reduction in productivity. Whilst, similar to saltmarshes and mudflats, seagrass beds can shift inland in response to sea level rise, they are then also threatened by coastal squeeze and a potential loss of supporting habitat in the correct tolerance range (MCCIP, 2018b). Seagrass beds are not considered to be sensitive to ocean acidification (MarLIN, 2019).

3.3.3. Native oyster habitat

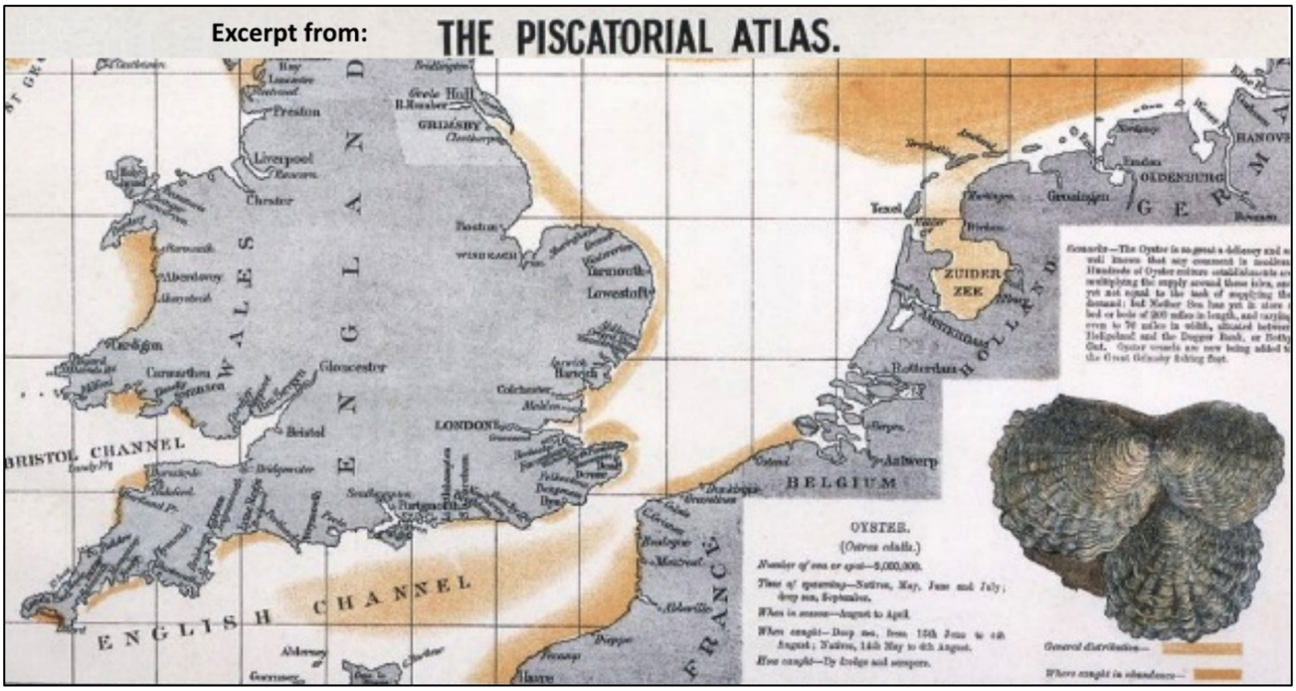
Welsh context

There are hardly any oyster habitats remaining in Wales, with only 0.01 km² having been mapped; the locations of which are considered sensitive, and hence are not discussed here (NRW, 2020).

Native oysters listed as a species of principal importance under Section 7 of the Environment (Wales) Act 2016. Oyster beds are also listed as a threatened and declining habitat under OSPAR.

Historical trends

Ostrea edulis has been harvested and cultivated in Europe since the Roman empire (Gunther, 1897). The Piscatorial Atlas of the North Sea from 1883 highlights very large areas as being occupied by oysters around the British Isles, including Wales (Image 6); noting that this map is considered to be indicative only. Given that the native oyster often resides in estuaries (see below), it is likely that many of the Welsh estuaries would have in the past also been home to oyster habitat, given the right conditions. Stocks of the bivalve declined throughout their entire geographical range due to over-exploitation and consequently low rates of recruitment, as well as declining water quality due to industrial and municipal effluents (Edwards, 1997, Mackenzie *et al.*, 1997, Tubbs, 1999). Episodic extremely cold winters in the 1960s and 1970s further diminished oyster beds, and over the past 40 years, the production of flat oysters has additionally been negatively affected by parasites and invasive species (Baud *et al.*, 1997; Harding, 1996; Utting and Spencer, 1992). Commercial landings of native oysters in Europe started to decline in the early 18th Century (Grizel and Heral, 1991). Today, as noted above, oyster beds have all but disappeared from Welsh waters, with only very small areas remaining.



Source: Olsen, 1883

Image 6. *Ostrea edulis* habitat indicated for the 19th Century

Ecology

The bivalve *O. edulis* (see Image 7) is found from the low intertidal shore down to sublittoral zones throughout the Atlantic and Mediterranean coasts of Europe. It is the UK's native oyster species. *O. edulis* is a protandrous alternating hermaphrodite species, meaning in its lifecycle it is first male and, when older, the oyster alternates between female and male functions (Laing *et al.*, 2005). In temperate UK waters oysters reach sexual maturity in the third summer after settlement (Kamphausen *et al.*, 2011; Korringa, 1952). Females brood fertilized eggs and larvae in their mantle cavity for 6 - 15 days, until the larvae have a fully formed shell of about 0.17 mm (Hedgecock *et al.*, 2007; Newkirk and Haley, 1982, Andrews, 1979; Orton, 1927). The larvae are then released into the water column.

Oysters usually spawn between late June and mid-September and remain dormant during winter; eggs or sperm are formed in spring (Hedgecock *et al.*, 2007; Kennedy and Roberts, 1999). When released into the water column larvae drift in the plankton for approximately 2 weeks. They then develop a “foot”, which enables them to settle on firm surfaces, followed by metamorphosis into a fully formed juvenile oyster (Laing *et al.*, 2005; Sobolewska and Beaumont, 2005).

The native oyster is associated with highly productive estuarine and shallow coastal water habitats (commonly 0 to 30 m depth) on firm bottoms of mud, rocks, muddy sand, muddy gravel with shells and hard silt. Although *O. edulis* have been shown to settle on a variety of substrates, research on their attachment preferences confirmed that they favour shells, oyster shells in particular (Korringa, 1946; Airoldi and Beck, 2007). Recruitment of *O. edulis* is sporadic and varies with environmental and physiological factors. Populations undergo natural phases of expansion and contraction. Successful recruitment appears to vary between one to 3 years (Loch Ryan, Scotland), or even every 6 to 8 years (Lough Foyle).

O. edulis are a key species that, at high enough densities, colonise areas of dead and living native oyster shells and form species-rich biogenic “reef” habitats. These habitats provide substrata, food, shelter and spawning grounds for a number of species (Smyth and Roberts, 2010), and support species including ascidians sponges, polychaetes, juvenile fish and crabs. Turf seaweeds are also likely to be present.

Image Credit: AER / Swansea University

Image 7. *Ostrea edulis* cemented to intertidal boulder

Sensitivity and tolerance

It is thought that natural *O. edulis* favour depths of 0 to 6 m and clean, hard substrates (specifically other oyster shells) to settle upon, although beds as deep as 50 m have been identified (Korringa, 1946; Airoldi and Beck, 2007). Native oysters are suspension feeders and hence require ample supply of suspended food particles, supplied to them on the currents. Oyster habitat is generally associated with very weak to weak (<1 knot) currents. While oysters tolerate a spectrum of environmental conditions, factors such as temperature, salinity, food availability and hydrodynamic conditions affect growth and morphology (Andrews, 1979). Reproduction is driven by temperature, and *O. edulis* require temperatures of 8 to 9°C to start growth (Korringa, 1957; Loosanoff, 1962; Wilson and Simons, 1985, Laing *et al.*, 2005). The species is able to survive in a wide range of salinities (18 to 40 psu), although low salinity may inhibit feeding. Similar to other bivalves, oysters inhale water and filter it through a gill chamber, thereby removing suspended food particles. Although oysters are adapted to turbid waters, high concentrations of suspended inorganic particles and sediment can result in reduced feeding efficiency, and thus growth (Grant *et al.*, 1990). Generally, native oyster beds are not considered to be highly sensitive to changes in water clarity or shading, deoxygenation, nutrient / organic enrichment or smothering / changes in siltation rate. They are however moderately sensitive to local decreases in salinity (MarLIN, 2020).

With regard to climate change, *O. edulis*, which occurs naturally from Norway to the Mediterranean, is considered to be fairly insensitive to most of the related pressures, including increases in temperatures, heatwaves, acidification and sea level rise (MarLIN, 2020).

3.3.4. Horse mussel beds

Welsh context

Approximately 8.7 km² of horse mussel (*Modiolus modiolus)* beds have been mapped in Wales (NRW, 2020). Although *M. modiolus* individuals are relatively common, *M. modiolus* beds (with typically 30% *M. modiolus* cover or more) are more limited in their distribution (NRW, 2019c). In Wales, a number of small separate beds have been recorded along the tide-swept coastline of north-west Anglesey (Rees, 2005). More extensive beds off the north side of the Llŷn Peninsula in Caernarfon Bay have been particularly well studied (e.g. Sanderson *et al.*, 2008; Rees *et al.*, 2008). It should be noted that, as horse mussels are a subtidal species, data collection relies on observational survey methods which are expensive and time-consuming. As such, it is possible that this habitat is more extensive than the area quoted above.

With regard to conservation importance, horse mussel beds are habitats of principal importance under Section 7 of the Environment (Wales) Act 2016. Horse mussel beds are also listed as a threatened and declining habitat under OSPAR.

Historical trends

As noted above, relatively small areas of horse mussel beds have been mapped in Wales, but there are large uncertainties with regard to their distribution given their relatively deep locations and consequent difficulties in mapping and surveying them. Whilst it is generally believed that beds have decreased in extent from historic baselines, amongst others due to their sensitivity to trawling and dredging (JNCC, 2008b), historic losses have not been estimated.

Ecology

Horse mussels are suspension-feeding bivalves which form beds in a range of environments, from tide swept channels to very sheltered areas. They resides in depths of 5 to 280 m (most commonly in the UK: 5 to 50 m). *M. modiolus* are adapted to live both semi-buried in sediment, as well as on the surface; in areas of strong currents and course material, the mussels bind sediment together and live infaunally (buried) in accreted sediment banks with a filament attachments to the substratum (Tyler-Walters, 2007). Epifaunally (on the surface), the mussels colonise hard substrates, such as bedrock, pebbles and shells, as well as the byssus / filament threads of other mussel species. Horse mussels significantly modify the underlying habitat, accruing faecal mud and shell debris over several years, and established *M. modiolus* reefs provide substratum and refuge for a for a wide variety of species including brittlestars, feather stars, crabs, whelks, sponges, sea firs, sea mats and sea squirts. The crevices between shells provide relief from physical and chemical stressors and create refuge from predation and competition, and are also important settling grounds for commercially important bivalve molluscs such as scallops.

While this species is adapted to a wide depth and exposure range, it is thought that growth rates of *M. modiolus* in areas of loose sediment / strong currents is reduced due to the increased byssus production necessary to maintain the integrity of the reef. Reefs in areas of strong currents will not form raised beds, as the strong tidal streams may prevent the faecal mud from being retained. Growth rates in deep water populations are also likely to be reduced due to a reduction in food availability (Comely, 1978). Mussels in areas with weaker currents are adapted to live epifaunally, and are sensitive to reductions in water flow rate, due to the consistent need for suspended particulate and phytoplankton supply present in well-mixed waters. Moreover, a decrease in already weak flow rates could expose the bed to increased siltation and suspended sediment accretion, resulting in interrupted feeding, poor shell growth and deoxygenation (Tyler-Walters, 2007). The degree of current speed may also influence the recruitment and survival of *M. modiolus* beds, with larvae in open areas likely to be swept away from the adult population. Such populations are probably not self-recruiting but dependant on recruitment from other areas, which is in turn dependant on the local hydrographic regime (Tyler-Walters, 2007).

Sensitivity and tolerance

As noted above, *M. modiolus* are adapted to survive in a range of environments, from moderately exposed to very sheltered, and are often found in the open coast, rockpools, straits, enclosed coasts and embayments (Tyler-Walters, 2007). *M. modiolus* are found in areas of very weak (<0.5 knots), weak (<1 knot), moderately strong (1-3 knots) and strong (3-6 knots) currents (when sediment accretion is above tolerable levels).

While there is limited literature on the effect of light penetration on *M. modiolus* beds, laboratory experiments conducted by Strömgren (1976) found an increase in growth rate both during and after periods of continuous darkness, suggesting a link between the pigmentation of the periostracum (the outermost layer of the shell) and light exposure. *M. modiolus* have a salinity range of 30 to 40 psu and are considered to be “highly sensitive” to decreases in salinity, although are likely to survive short-term exposure to reduced salinity conditions (Tyler-Walters, 2007).

M. modiolus are resistant to changes in oxygenation, such that they are tolerant of hypoxia and exposure to hydrogen sulphide, and exhibit anaerobic metabolism to some degree. Research furthermore suggests that the species is tolerant of high nutrient levels (Richardson *et al.*, 2001; Tyler-Walters, 2007). Moderate nutrient enrichment may be beneficial by increasing phytoplankton productivity and organic particulates, and hence food availability (Tyler-Walters, 2007). Eutrophication however may have negative effects on *M. modiolus* beds, causing increased turbidity, deoxygenation and algal blooms (leading to an accumulation of paralytic shellfish poisoning (PSP) toxins (Shumway, 1990; cited in Tyler-Walters, 2007)). Temperature wise, no specific temperature ranges are noted in the literature, but it is important to realise that *M. modiolus* is a boreal (northerly distributed) species reaching its southern limit in British waters (Holt *et al.*, 1998). They are thus considered to have a high sensitivity to increases in temperature (MarLIN, 2007).

Horse mussel beds in Wales are potentially threatened by several climate change stressors, including rising seawater temperatures, ocean acidification, changes in wave exposure and ocean currents. Horse mussels exhibit many characteristics which make adaptation to changing conditions difficult. This includes late reproductive maturity (5-6 years), low larval settlement success and a sporadic reproductive output. Thus, their sensitivity to climate change impacts, particularly temperature changes, may be particularly high (MCCIP, 2018c).

3.3.5. Sabellaria alveolata reefs

Welsh context

Just over 5 km^2 of *S. alveolata* reef have been mapped in Wales, with notable extents present around the Lleyn Peninsula and Cardigan Bay, was well as in the Severn Estuary and wider Bristol Channel area, as well as the (NRW, 2019d).

S. alveolata reefs are a habitat of principal importance under Section 7 of the Environment (Wales) Act 2016. S. alveolata reefs can be considered as biogenic reefs, and are encompassed by the following two Annex I habitats: reef and large shallow inlets and bays. *S. alveolata* reefs in at least three Welsh SACs contributed to ‘reef’ features being included in their designation, namely the Lleyn Peninsula and the Sarnau SAC, Cardigan Bay SAC and the Severn Estuary SAC. *S. alveolata* reefs are furthermore a designated feature of a number of SSSIs in Wales.

Historical trends

JNCC (2008c) reported that there had been historical contractions in *S. alveolata* reef range in a number of areas in the UK, including the upper parts of the Bristol Channel and in the Dee Estuary. Causes had not been postulated and it was considered difficult to assess the true significance of these changes given the natural variability of the species. NRW (2019d) however note that overall, the extent and distribution of *S. alveolata* reefs in Wales ‘are thought to be increasing’.

Ecology

The honeycomb worm *S. alveolata* is a sedentary polychaete worm that constructs tubes in tightly packed masses with a distinctive honeycomb-like appearance. These reefs can be up to 50 cm thick and take the form of hummocks, sheets or more massive formations, consisting of coarse sand grains and shell fragments . It is one of the most prolific reef builders and colonises lower to mid intertidal shores, but can be found subtidally, as it is in the extensive Severn Estuary reef (Mettam *et al.*, 1994).

S. alveolata require firm attachment surfaces and colonise different types of hard substrata, including bedrock, boulders / pebbles, large bivalve shells and anthropogenically created hard structures such as rock revetments even tires and boats (see Image 8), settling mainly on or near established tube structures or degraded reef scars (Firth *et al.*, 2015). They have also been known to settle on sediment that has been stabilised, such as by the sand mason worm *Lanice conchilega* (Larsonneur, 1994).

Reefs of *S. alveolata* create topographic and ecological niches for other species; the tube-reefs reduce physical and chemical stresses for intertidal species, create refuge from predation and competition and even alter resource availability. In intertidal reefs, seaweeds including fucoids, red and green algae, are often recorded, as well as animals including barnacles, dogwhelks, winkles, mussels, small crabs and other bivalves.

Source: AER / Swansea University

Image 8. *Sabellaria alveolata* colonising artificial material in Welsh intertidal areas

Sensitivity and tolerance

S. alveolata is a filter feeder and requires suspended food particles, as well as ample sediment particles in order to construct the reef formations. It is therefore associated with exposed coastal conditions and strong to moderate wave action / turbulent high current velocity waters which transport sufficient food and sediment. The reproductive biology of *S. alveolata* also indicates the importance of the hydrodynamic regime for dispersal and recruitment, as the pelagic planktonic larvae settle only after six weeks to six months. Therefore, depending on local conditions, they may remain within a discrete area and colonise neighbouring reefs (Ayata *et al.*, 2009).

Temperature affects the growth and mortality of *S. alveolata*, such that the metabolism of the species, and the associated growth, of *S. alveolata* increases with temperature, with a plateau at 20°C. Below 5°C, growth becomes constrained; worms often die during prolonged periods of low temperatures (Egerton, 2014) and mass mortality happen during exceptionally cold winters (Holt *et al.*, 1998).

While little is known about the influence of light on *S. alveolata* reefs, the presence of photoreceptors on the specialised anterior end of *S. alveolata* individuals suggests a certain tolerance to a range of light exposures. *S. alveolata* reefs are reported to be resilient when faced with nutrient (organic and inorganic) enrichment, local temperature fluctuations (increases and decreases), local salinity increases and local changes to water flow and wave exposure. They are moderately sensitive to local decreases in salinity (preferring fully saline conditions between 30 and 40 psu), changes in suspended solids / water clarity, penetration or disturbance of the substrate and smothering / siltation rate changes (MarLIN, 2008).

No specific literature on the resilience of *S. alveolata* to climate change could be found. However, this species can be found from the Mediterranean to the north Atlantic, and the British Isles form the northern limits of the distribution in the north east Atlantic; they are thus likely to be relatively tolerant of slow in increases in sea temperature. Given the sensitivities reported in MarLIN (2008), this species will likely be low to moderately sensitive to sea level rise, increases in storminess, and changes in rainfall pattern.

3.4 Key Mechanisms / Techniques for Restoration

For each of the six habitats which are the focus of this report, key restoration techniques are now outlined.

3.4.1. Saltmarsh and mudflat restoration

Three main techniques have been identified for the creation and restoration of intertidal mudflat and saltmarshes. These include:

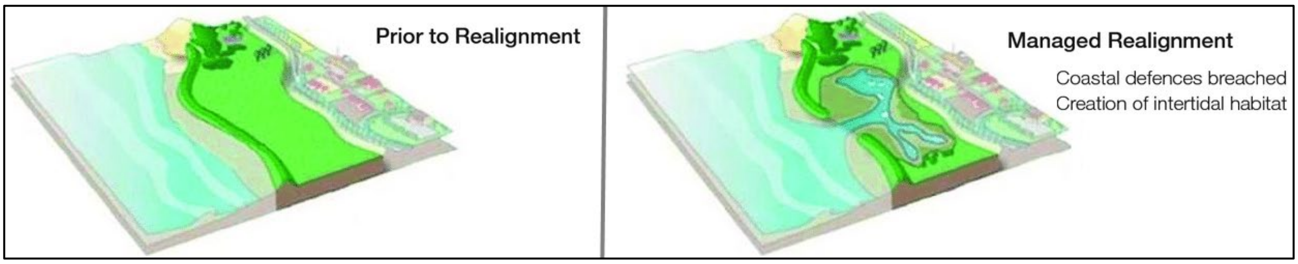
- Managed realignment (including regulated tidal exchange (RTE));
- Beneficial use of dredged sediment / Sediment recharge; and
- Manipulation of natural processes (encompassing sedimentation polders etc.).

Hybrids of these techniques are also feasible, and have indeed been implemented.

Please note that two relevant UK guidebooks (on saltmarsh restoration and beneficial use of dredged sediment) are currently being produced and are due for publication by the summer of 2021. These, and other available guides (notable Leggett *et al.*, 2004 and Nottage and Robertson, 2005) should be consulted by those wishing to undertake saltmarsh and / or mudflat restoration.

Managed realignment

Managed realignment is generally viewed as the main option for the creation of intertidal habitats, but it can also be used to create subtidal habitat in low lying areas, or in combination with sediment reprofiling. It involves the deliberate breaching, or removal, of existing seawalls, embankments or dikes in order to allow the waters of adjacent coasts, estuaries or rivers to inundate the land behind (see Image 9).



Source: Zhu *et al.*, 2010

Image 9. Managed realignment diagram

Regulated tidal exchange (RTE) is a form of managed realignment / intertidal habitat creation that allows the controlled inundation of defended land by saline water through the use of weirs, sluices, culverts and / or pipes inserted into a flood protection embankment. RTE differs from managed realignment in that the sea wall remains intact. Furthermore, through the use of tidal exchange mediums such as sluices and culverts a high degree of control is retained, the tidal flow and water exchange volumes are restricted and the existing defence line tends to require continued maintenance, and in some cases, upgrades. In most instances, the newly flooded land is low-lying coastal floodplain and therefore a new seawall is needed to clearly define the inundated area and protect the hinterland behind. However, on areas with rising ground either no new line of defences or only a partial counterwall is required.

To date, including RTEs, over 120 managed realignment schemes have been implemented across Northern Europe, 75 of these are in the UK, with four in Wales (ABPmer, 2020), including schemes at Cwm Ivy and Ynys Hir (see Image 10). Around a third of the UK schemes (and one in Wales) were primarily motivated by compensation requirements, including in relation to port developments and flood defence schemes or strategies.

Source: OMReg (https://www.omreg.net/); outlines copyright ABPmer

Image 10. Ynys Hir (Dyfi (Ceredigion)) and Cwm Ivy managed realignments (Loughor (West Glamorgan / Swansea))

The UK schemes were generally implemented on uninhabited agricultural floodplain land without significant existing infrastructure or nature conservation designations (though the fronting estuarine habitats have frequently been highly designated). Many of these areas would have previously been intertidal habitats, having been claimed decades or centuries earlier.

Evidence from implemented schemes suggests that these have been successful in establishing intertidal habitats, and have tended to show rapid ecological development in terms of supporting invertebrates, birds and establishing saltmarsh. However, it is important to ensure that such sites are designed appropriately, notably in relation to having the correct elevation to deliver certain habitat types and having appropriately designed drainage channels and creeks to enhance saltmarsh development and fish usage. Past projects have also shown the value of maximising the degree of on-site morphological complexity to create multiple ecological niches and enhance the level of biodiversity achieved (ABPmer, 2017).

Managed realignment can be especially valuable for saltmarsh creation, though functional equivalency with adjacent mature marshes can take several years to achieve (e.g. Brown *et al.*, 2007). Mudflat creation has also been successfully achieved in many cases, though in estuaries with a high sediment load, such as the Severn, rapid accretion has occurred, elevating significant proportions of managed realignment sites out of the mudflat range after a few years. However, in estuaries with lower sediment loads, accretion rates over mudflats tend to be significantly lower, and mudflat can thus be expected to be maintained for several decades; for example, at Allfleet's Marsh on Wallasea Island (Essex, England), around 95 % of the original mudflat extent is still retained 14 years post breach (ABPmer, 2020). Saline lagoons, (bird) islands, transitional and 'terrestrial' habitats are also frequently included within managed realignment boundaries, and RTEs are particularly useful for establishing these habitats.

Sediment recharge / beneficial use of dredged sediment

Sediment recharge in intertidal areas is a process by which dredged sediments are placed over or around intertidal mudflats and saltmarshes to either create habitat (most often saltmarshes), or restore or protect intertidal habitats from ongoing erosion (Nottage and Robertson, 2005; Defra and Environment Agency, 2007). This approach is particularly valuable for protecting habitats that are sediment starved or subject to erosion and where the introduction of dredge arisings will allow the habitat to cope with, or respond to, sea level rise.

In the UK, approximately 20 intertidal recharge projects have been undertaken to date; some of which recur on a regular basis. These have all been in England, with many projects in Essex, Suffolk and on the South Coast. Such projects can differ greatly in scale (i.e. the area of deposition or the volume of sediment used), and on the basis of the number and type of structures, if any, that might be put in place to retain sediments once they are deposited. In simple terms, the following five approaches represent the main ways in which dredged sediment has in the past been placed directly onto intertidal habitat (ABPmer, 2017):

- Back-hoe extraction translocated for back-hoe placement;
- Back-hoe extraction translocated for pumped placement via pipe;
- Back-hoe extraction translocated for intertidal bottom dumping;
- Suction dredge with direct pumped placement; and
- Suction dredge translocated for pump / rainbow release.

In many estuaries in the UK, fine materials dredged during maintenance and capital dredging campaigns are deposited in a subtidal location within the same estuary; not to create mudflat from subtidal, but to essentially trickle charge sediment back into the estuarine system. This is, for example, practiced in the Severn Estuary, where Cardiff Grounds and Newport Deep are key subtidal deposit grounds which lie a few miles from the dredge locations. The hypothesis behind this approach is that there is a net balance between the amount of material being deposited and eroded in many tidal estuaries. Such a balance may be disturbed when an estuary is dredged, and continuous permanent removal of materials could eventually lead to erosion of intertidal habitats (Cefas, 2009).

The direct placement of material onto the subtidal in order to elevate an area into the intertidal, and thus create mudflat, has never been practiced in the UK. There have however been examples of this in the US and Japan, where recharge has been very widely practiced for decades (PIANC, 2009).

It is worth noting that there has been a long-term desire to see more dredge arisings used beneficially for environmental and / or socio-economical activities, at a national and international level. For example, the need to seek beneficial use opportunities was identified within the 1996 International Maritime Organisation (IMO) London Protocol and other dredge management reviews and guidance (HELCOM, 2015). The WNMP also contains a narrative noting that 'the beneficial use of dredged material is encouraged'. Many organisations / studies over many years have described this situation (Royal Society for the Protection of Birds (RSPB), 2018; Permanent International Association of Navigation Congresses (PIANC), 2009). The reasons why there is limited beneficial use are therefore well-rehearsed, and can be summarised as follows:

- Technical difficulties of achieving projects and the resulting extra costs incurred;
- Extra tasks that are required to progress through consenting processes;
- Lack of bespoke legislation or single government body / department to champion this subject;
- Absence of any transparent mechanism for identifying sediment sources and linking these locations with sites that have a need for sediment; and
- Concerns regarding the environmental effects that could arise from the beneficial use activity.

In essence these factors mean that it is almost always easier to maintain long-established 'business as usual' practices and place sediment at an established disposal site rather than go through a new and lengthy process to undertake a beneficial use project. Of all these issues, the fundamental ones are that it typically costs more money and also that those who incur the costs (the dredging operators, marina owners and harbour authorities) are not those that benefit and so there is a fundamental disincentive for beneficial use measures to be pursued.

Manipulation of natural processes

The manipulation of natural processes encompasses projects which alter the existing sedimentary regime along a shoreline in order to protect habitat and possibly create mudflat and saltmarsh. This includes a wide range of possible techniques such as introducing obstructions or altering shorelines. Such structures are installed in areas which are exposed to relatively high tidal or wave energy forces which would normally prevent the settling of sediments, or re-suspend any that had settled during slack periods. This is provided that the suspended sediment concentration in the system is high enough for accretion to take place. Thus, the artificial import of sediment is not necessary, but instead, structures are put in place to reduce energy and encourage sediments to settle and accrete. In the UK, to date, these have generally focussed on saltmarsh erosion protection and mudflat accretion encouragement.

There are techniques such as shore perpendicular groynes, which can potentially be used to expand mudflat seawards, onto existing subtidal areas, though there are no known (intentional) examples of this in the UK, with success generally very much dependent on local conditions, notably sediment loads.

In the past, the main methods used for intentionally increasing sedimentation in intertidal areas have included brushwood fencing, polders / sedimentation fields, wave breaks or groynes. Wavebreaks are generally located some distance offshore in more exposed situations, usually in parallel to the shoreline. Sedimentation fields / polders have been used extensively along the Dutch and German Wadden Sea coasts for centuries, and there is one Welsh example of its use near Cardiff, at Rumney Great Wharf (Image 11).

Taken by NRW, 2005

Image 11. Picture of Rumney Great Wharf polder fencing under construction

Land claim would have traditionally been the ultimate aim of this method; although more recently it is undertaken to build up saltmarsh in front of coastal defences. This technique was trialled in at least 17 locations in Essex in the 1980s, with mixed success (French, 2001). A recent trial of installing coir roll structures along eroded saltmarsh creeks by the Essex Wildlife Trust (EWT) has shown some promise in trapping sediment and thus restoring marshes in the Blackwater Estuary (EWT, 2021). At Rumney Great Wharf, it is believed that the five polders, which were installed in 1999 and 2005, had some success, but this was limited due to the polders not having been maintained (or monitored). This meant that the brushwood bundles had been washed out within a few years of the polders' installation, and many of the fence posts damaged or washed away. Repair works are currently being considered by NRW (NRW, 2021a).

Costs of mudflat and saltmarsh restoration techniques

Collectively, there is now a comprehensive evidence base for most of the restoration techniques, providing a robust practical technical foundation for future saltmarsh and mudflat restoration schemes. Experience has however shown that implementing such projects often requires a lot of preparatory analysis, consultation, planning and assessment work and they can be costly, especially at a large scale.

The cost incurred for habitat restoration through RTE and managed realignment is dependent on the scale and location of the work needing to be undertaken, as well as the extent of engineering work and potential ongoing intervention. As such, managed realignment scheme costs are highly variable, with those for implemented UK schemes having ranged from £850 to £156,000 ha^{-1}. The average managed realignment unit costs are approximately £41,000 ha^{-1} (ABPmer, 2017; values converted to 2020 prices), with costs of compensatory schemes generally being at least twice as expensive. However, it should be noted that managed realignment projects generally lead to long-term cost savings in terms of flood risk management, particularly where a given embankment had been in a poor state of repair.

Beneficial use (of dredged sediments) costs are typically quoted in costs per unit of sediment deposited. These can range very widely, largely depending on the amount of double handling and preparation required. Costs have ranged between £8 m^{-3} to £122 m^{-3} in the UK (NRW, 2019e). The highest cost quoted here was for the Wightlink Boiler Marsh saltmarsh restoration project at Lymington in Hampshire. This required double handling, whereby the materials were unloaded onto a purpose-built barge / platform, then water added to the materials to facilitate pumping (the materials had been dredged using a backhoe, so were too solid). Long pipes led from the platform to the degraded saltmarsh, where 10 poldered fences with 3 m high stakes had been installed, with hay bales inlaid into and around the fences. These fences were installed to ensure that most of the sediment sludge, once pumped, would remain on the marsh, rather than running off into the Solent (retention fencing or similar is important with pumped recharge schemes) (ABPmer, 2018).

A relatively rare example of a nearly cost neutral beneficial use project is Lymington Harbour Commissioner’s (LHC) Boiler Marsh scheme. This recharges the same marsh complex as the Wightlink project quoted above, but with a different technique, and at mudflat elevations. Here, materials extracted using back hoes is then routinely bottom dumped on low mudflats in an embayment a short distance from the dredge locations. LHC’s average costs have reduced from £9.80 m^{-3} during the trial phase to £8.70 m^{-3} for deposition over the following two winters. These average costs are slightly lower than the costs of taking this material to the disposal ground (which is £8.78 m^{-3} on average). This reduction has largely been due to a reduction in monitoring costs / effort. It is also worth noting that, due to the relatively compacted nature of the backhoed materials, around half of the materials have persisted, and that a clearly raised high mudflat feature which measures around 1.4 ha, has become established (ABPmer, 2019). In addition, the dredging vessels are quite small and are able to work with draughts of less than 3 m. This facilitates an efficient disposal process without the expensive need for pumping or double handling, but it does mean that the sediment does not immediately benefit saltmarshes, as these are too high for the vessels to reach. Pumping materials directly onto saltmarshes tends to be more complicated and expensive. For example, two other beneficial use projects undertaken at Lymington, which utilised the pumping technique, had volume costs of 32 m^{-3} and 122 m^{-3}. The latter quoted cost was for the Wightlink compensation project at Boiler Marsh, which cost just over £500,000 for two campaigns aiming to restore 1 ha of saltmarsh (this is the most expensive beneficial use project in the UK to date) (ABPmer, 2019).

Sedimentation polders can also be relatively expensive to install, and require regular inspection and maintenance. For example, the installation of four polders at Rumney Great Wharf in 2005, cost £190,000 (£286,000 in 2019 prices, or £71,000 per polder and some £26,000 per hectare of polder area). Depending on the nature of the location (e.g. how exposed it is to wind-generated waves), maintenance costs may be high. In Germany, the federal state of Lower Saxony spends about £1 million per year on polder and saltmarsh maintenance along 70 km of mainland coastline. This includes repair of brushwood fences, but also the repair of the dyke toe protection and the maintenance of ditches, and is equivalent to ~£15 per metre (Dornbusch, 2019).

3.4.2. Seagrass bed restoration

Seagrass restoration has been conducted for over 50 years globally, and the means of doing so can principally be split into three major techniques:

- Replanting;
- Reseeding; and
- Natural recolonisation.

Please note that a relevant UK guidebook is currently being produced, and is due for publication by the summer of 2021. This should be consulted by those wishing to undertake seagrass restoration.

Replanting and Reseeding

Both replanting and reseeding to achieve seagrass restoration have their relative merits and have exhibited varying levels of success. A broad overview of the literature illustrates that, although a lot is now known about seagrass restoration, much more remains to be researched and, as a result, the success rate of restoration projects is still often very low.

Adult shoot replanting usually involves harvesting plants from an existing meadow and transplanting them to the restoration site once suitable conditions have been established for seagrass survival. Otherwise aquaria or nursery-grown seagrasses are used for translocation. The use of reseeding generally relates to the collection and targeted redistribution (and sometimes processing) of wild seeds. Researchers in Great South Bay, New York developed a method of seeding which involved attaching seeds to a biodegradable tape. The tape is then planted just below the sediment surface at the desired restoration site (Churchill *et al.,* 1978).

Replanting of seagrasses uses either labour-intensive diving techniques or various mechanistic approaches to planting various sizes and ages of seagrass plants into new localities. In the US, reseeding and replanting techniques have sometimes been used together and, using seeds possibly in conjunction with adult plants may in some instances prove more effective (van Katwijk *et al.*, 2016).

In most cases, some means of anchoring the shoots to the bottom is necessary until the roots can take hold. One such example of this is the 'Save The Bay' restoration project for Narragansett Bay (New Hampshire) in the US; this has used a specialised remote transplant methodology known as "Transplanting Eelgrass Remotely with Frames" (TERF). The TERF method involves using clusters of plants temporarily tied with degradable crepe paper to a weighted frame of wire mesh. This method has resulted in relatively a high survival rates (58.7-69.0%) of transplanted eelgrasses in other studies in Korea. The "Shell Method" is another successful eelgrass transplanting method in which oyster shells are used as an anchoring device, and does not require diving for subtidal transplanting (Park and Lee, 2007). Many other forms of planting methods including seeding, stapling, use of anchored and unanchored sprigs, plugs, peat pots, and transplanting of individual mature plants have been trialled at various locations (Phillips, 1980; Fonseca, 1994; Fonseca *et al.,* 1998), with varying success. Fertilization of transplants to accelerate growth and bed coalescence has also been trialled, however the benefits in eelgrass restoration projects have been inconclusive (Fonseca, 1994).

Seagrass restoration projects are unfortunately often not successful. Historically, failures have often been due to suboptimal consideration of the habitat requirements for seagrass and the continued presence of the stressor that caused the original seagrass loss (e.g. eutrophication). A recent global review study also highlights the need for restoration to occur at sufficient scales in order to facilitate positive feedbacks and to spread the chances of success (van Katwijk *et al.*, 2016). With regard to techniques, seeds, adult plants and intact units of native sediment with roots (sods) were not found to be significantly different, although seedlings showed lesser planting results. A short distance to the donor site was also related to success. Whereas transplantations (replanting) frequently fail (60%) or have limited success, a substantial number of transplantations showed substantial expansion rates (van Katwijk *et al.*, 2016).

Bos and Katwijk (2005) describe attempts by the Dutch authorities to reintroduce seagrass to create a stable population in the Dutch Waddenzee. The rationale behind the programme was to create a source stock for further recovery and expansion along the coast. Site selection was considered to be highly important, with locations chosen using the following criteria (Reach *et al.*, 2015):

- Areas where *Z. marina* was known to have been present / grown naturally in the past;
- The area should have natural protection against prevailing winds;
- The area should have some freshwater input; and
- No fishing activities, or bait digging, should be allowed in, or within proximity of, the area.

In the UK, many previous seagrass restoration trials also failed historically. However, several initiatives are currently ongoing which are showing promising initial results. In Wales, Project Seagrass, as part of the Seagrass Ocean Rescue (with WWF, Sky Ocean Rescue, Swansea and Cardiff Universities), are restoring seagrass near Dale, West Wales, using a technique whereby hessian bags filled with seeds are secured on the seabed. The restoration process has been done in collaboration with local people, students and volunteers (using more than 1 million seeds in the process). In November

2020, the last seed at the 2 ha site was planted, and first shoots from earlier plantings were successfully observed in the summer of 2020 (Project Seagrass, 2020).

Natural Recolonisation

Seagrass restoration through natural recolonisation projects tend to focus mainly on water quality improvement in a given study area, with the assumption that once suitable conditions are established, seagrass will naturally (re-)colonise. This approach can involve a coordinated effort to upgrade sewage systems, and programmes to identify and curtail point and non-point discharges from industrial, residential and agricultural areas in the coastal zone. Such water quality improvements may be factored into other restoration techniques in order to increase the longevity of a successful restoration. Many restoration initiatives furthermore target the installation of less damaging boat moorings, as existing chain moorings can damage seagrass meadows. The English ReMEDIES project is for example investigating this, as well as seeding initiatives, mainly around Plymouth Sound in Devon (having been launched in early 2020) (ReMEDIES, 2020).

Costs of seagrass restoration techniques

Seagrass restoration has the capacity to be both very expensive and (as noted above) has a high risk of project failure, with seagrass restoration costs higher than terrestrial plant restoration (Kenworthy *et al.,* 2018). Bayraktarov (2016) quote median to average per-hectare costs of between £88,000 and £322,000 for seagrass restoration (2020 prices).

3.4.3. Native oyster (*Ostrea edulis*) restoration

Oysters are one of the most commercially attractive marine species; this has motivated restoration efforts throughout Europe for centuries. Restoration techniques can be summarised as follows:

- (Re-) laying of adult oysters;
- (Re-) laying of spat (very young oysters); and
- Provision of shell cultch (substratum for larvae to settle) directly on the seabed.

A UK guidebook has recently been published with respect to oyster restoration (Preston *et al.*, 2020). The UK and Ireland Native Oyster Network is also worth noting, this is a community of academics, conservationists, oystermen and NGOs who are working to restore self-sustaining populations of native oysters.

Provision of cultch / habitat

As noted in Section 3.3. above, on oyster habitats, settlement surfaces include the shells of living and dead oysters, other shellfish and other hard substrata such as stones and wood. In managed fisheries, old bivalve shells are often added as cultch to encourage native oyster settlement, with oyster, scallop and mussel shells reportedly providing particularly viable surfaces (Laing *et al.*, 2005; Key and Davidson, 1981). Where suitable settlement materials are not present, then they will need to be introduced. Preston *et al.* (2020) note that '*finding the optimal settlement substrate is a trade-off between availability, price and aim of the project'*. Apparently, '*good settlement can be achieved on both stone aggregate and various shell types*', but shells should be purchased well in advance (ideally at least 12 months) to allow for weathering and / or other biosecurity treatments. In addition, depending on location, a period of stabilisation may be necessary for the deployed materials (Preston *et al.*, 2020).

Laying of oysters

While maintaining or enhancing seabed habitats to support spat settlement is an essential starting point for native oyster restoration, a viable broodstock must also be present if the spat are to be produced in the first place (Reach *et al.*, 2015). Where there are few adults, or where the adults are too dispersed, regular successful spawning may be impaired and broodstock enhancement may thus be needed. Where available, this enhancement could take the form of aggregating adults by collecting them from the wild and depositing them together in specific locations. This is however often an unviable option in the UK, due to impacts on the source location. Alternatively, when available, broodstock may be sourced from hatchery stocks, with dedicated hatcheries sometimes established for restoration projects (Preston *et al.*, 2020). Both approaches have merit and carry risks; for example moving wild shellfish from one site to another increases the risk of introducing disease or non-native species to an area. Indeed, such movement is often prevented by legislation in relation to the *Bonamia* parasite. In addition, reliance on hatchery sourcing also increases the risk of limiting the genetic diversity of the population (Laing *et al.*, 2005).

General considerations

There are many biotic, abiotic and socio-economic factors to consider when planning the restoration of a native oyster habitat, including but not limited to, site selection, human pressures, substrata, environmental conditions, disease (notably the parasites such as *Marteilia* and *Bonamia ostreae* (Pogoda *et al.,* 2019; Pogoda, 2019), protection from unregulated fishing and larval availability. Ideally, oyster restoration should occur in areas less impacted by human activities, and away from other commercial species which may be targeted by bottom trawlers (Cook *et al.*, 2013). An assessment would have to be made as to whether sufficient natural settlement substrate for oyster larvae was present as otherwise, even in waters with sufficient larval abundances, oyster populations may not recover / establish (Pogoda, 2019). Recruitment of *O. edulis* is sporadic and varies with environmental and physiological factors, with populations undergoing natural phases of expansion and contraction, making the success of a restoration project hard to quantify. The Native Oyster Restoration Alliance (NORA) have six recommendations for oyster restoration:

1) Produce sufficient oysters for restoration of oyster reefs; support existing hatcheries, spatting ponds and spat collector techniques and to establish new hatcheries and spatting ponds for the production of robust and genetically diverse *O. edulis* seed. Broodstock sanctuaries should be established and used for local reinforcements.
2) Identify and create suitable sites for restoration of oyster reefs; sufficient undisturbed and suitable areas should be identified for the restoration and protection of *O. edulis* in all regions of its indigenous range. This includes the restoration of suitable substrate in some areas and includes “reintroduction” sites where *O. edulis* was previously recorded, “reinforcement” sites where *O. edulis* is present in very low densities and “conservation” sites where *O. edulis* is “abundant with sufficient reproduction and settlement for the habitat to persist in the long term”.
3) Provide suitable substrate for successful recruitment (such as *O. edulis* shells) with the extraction of *O. edulis* shells from the marine environment for other usages becoming prohibited within the restoration area.

4) Respect *Bonamia*-free areas; research and understand the biology of the *Bonamia* parasite and infection dynamics, and strictly follow biosecurity protocols in *Bonamia*-free areas.
5) Create common monitoring protocols that will provide comparable results for projects throughout Europe and for restored sites should be developed and followed. Where possible, monitoring should include the assessment of ecosystem services on a habitat and ecosystem scale.
6) Preserve genetic diversity; established hatchery and pond production protocols should be adapted to preserve the extant genetic diversity of native oysters in Europe.

UK and Irish restoration schemes

Various attempts at oyster restoration have taken place over the last 50 years, with varying short and long term successes. In Ireland, bed rotation has been trialled, i.e. the collection of spat on cultch for seeding, which is then transferred to other suitable areas. In 1991, 250 bags with 1,000 spat each on native oyster cultch from Tralee Bay were transplanted to Lough Swilly and grown on trestles for over a year. They were subsequently seeded onto the seabed. These projects increased the catch rates the following years, but limited spat supply and the challenge of relocating oysters whilst minimising the spread of diseases were significant challenges (OSPAR, 2009).

In 1997, an oyster relaying project in Strangford Lough, Northern Ireland was initiated, in collaboration with the local fishing community (Kennedy and Roberts, 2001, Laing *et al.*, 2005). Cultch, seeds and adult oysters were placed at nine sites. The oyster population increased from 100,000 individuals in 1998 to 1.2 million individuals in 2003, but stock levels were not sustained due to unregulated harvesting and infestation by the *Bonamia Ostreae* parasite, leading to a decline to 650,000 individuals by 2005 (Smyth *et al.*, 2009).

In 2010, the Chichester Harbour Oyster Partnership Initiative was established, which involved the relaying of 2,298 kg of broodstock oysters on the sea bed at a density of 40 m^{-2}. Oysters reproduced successfully up until the spawning of larvae, but the sex ratio (male: female) of the broodstock was 3:1, differing significantly to what was naturally expected (1:1). Two years after relaying, an increased mortality of the relayed oysters was reported. It was concluded that the environmental conditions at the seabed might have negatively affected oyster physiology, reducing growth and leading to increased mortality (Eagling, 2012).

Other recent initiatives are also noteworthy. These include the Essex Native Oyster Restoration Initiative, whereby more than 25,000 mature native oysters were re-laid in the Blackwater, Crouch, Roach and Colne estuaries in 2016 (EWT, 2016). A recently initiated Scottish trial is also of interest. In the Dornoch Firth, work began in 2018 to restore oyster reefs which were fished to extinction over 100 years. Here, shell cultch has been provided, and about 20,000 oysters were initially placed on this in a grid formation. The aim is for the reefs to become self-sufficient and sustain 4 million oysters in a 40 ha area (BBC, 2018).

Lastly, two Welsh projects are currently under way. Firstly, in Milford Haven, an NRW-led initiative is trialling the introduction of cultch and aquaculture reared juvenile oysters across two sites (NRW, 2021b). Secondly, as part of the UK Wild Oysters Project, caged oyster systems, or nurseries, will soon be installed in Conwy Bay (Native Oyster Restoration Alliance, 2020).

Costs of native oyster restoration techniques

Oyster restoration initiatives tend to be costly. The Dornoch Firth project for example involves the investment of £6.4 million in order to restore 40 ha. £1.4m of this is being spent on sourcing native oysters and setting up a shellfish supply chain. Overall unit costs for the Durnoch Firth scheme are around £160,000 per hectare. Preston *et al.* (2020) provide an Australian case example, whereby 20 ha of Australian native oysters were restored in 2015 at a cost of £1.9 million (£2.1 million in 2020 costs, equating to £105,000 per hectare). Bayraktarov (2016) calculated median to average per-hectare costs of between £28,000 and £329,000 for oyster bed restoration (2010 US dollar costs converted to pounds Sterling and 2020 prices; based on 23 projects).

3.4.4. Horse mussel (*Modiolus modiolus*) restoration

While horse mussel restoration projects are few, experimental trials suggest the translocation of adult horse mussels has the potential to accelerate reef recovery (Roberts *et al.*, 2011; Elsäßer *et al.,* 2013) and has therefore been proposed as a suitable restoration technique. Experimental variables incorporated within trials to date have included substrata, elevation, current flow and larval dispersal to try and determine the most suitable method of *M. modiolus* restoration.

Most notably, a comprehensive study by Roberts *et al.* (2011) investigated potential intervention strategies to restore *M. modiolus* beds in the Strangford Lough inlet in Northern Ireland, and studied the effects of

a) translocation of *M. modiolus* onto artificial reefs (comprising of scallop (*Pecten maximus*) cultch) of varying elevations;
b) translocation of *M. modiolus* directly on the seafloor;
c) using potentially suitable substrata (*M. modiolus, M. Modiolus* shells and *P. maximus* shells) for spat settlement; and
d) testing a pilot *M. modiolus* hatchery cultivation on M. modiolus recruitment.

For both translocation options (a and b), Roberts *et al.* (2011) found that individual *M. modiolus* clumped fast and remained attached to all substrata and that survival rates were high. Whilst elevated plots initially offered more protection against mobile predators, mussel survival was not significantly better on elevated cultch. Numbers of species associated with the constructed reef increased with greater habitat complexity of the reef structure, indicative of a natural reef-forming process by *M. modiolus*. It was concluded that mussels translocated to artificial reefs stabilise quickly, show high survival, and are rapidly colonised by other organisms.

The spat settlement study (c) found this was very poor on artificial spat collectors and loose shells, but was significantly better among clumps of live mussels than on other materials. Study d) found that larvae preferred settling among live mussels to artificial substrata, and that the high costs associated with running hatchery operations compared to a poor return in seed made this option unviable (Roberts *et al.*, 2011).

Some of the translocation trials which have taken place to date saw the spread of horse mussels beyond the original translocation sites (Levin, 2006; cited in Elsäßer *et al.*, 2013). This is due to larvae potentially travelling quite long distances in areas where current speeds are high. A study by Mackenzie *et al.* (2018) found that horse mussel bed populations from semi-enclosed water bodies (e.g. sea lochs and firths) appeared to act as a source of migrants rather than a sink from adjacent populations.

With regard to Stranford Lough, a 2014 study determined that the restoration sites which were created in 2010 had in fact failed, and no obvious *M. modiolus* reef remained at the site four years on (Geraldi *et al.*, 2014). Consequently, and also due to concerns over impacts on the donor site, the Northern Irish authorities decided against upscaling earlier translocation trials and to instead focus on protection. Thus, earlier protection measures were strengthened, such that now, fishing using trawling and dreding methods is banned throughout the Lough, and potting is subject to permitting. Several Sea Fishing Exclusion Zones exist where mobile and static gear fishing is completely banned. There are also limits on mooring, anchoring and diving, and fishing officer and warden posts have been created. Lastly, populations sited just outside the Lough, but which are considered to facilitate larval recruitement for the Lough itself, have been afforded protection (Department of the Environment and Department of Agriculture and Rural Development (DAERA), (Northern Ireland), 2015). Monitoring surveys undertaken in 2019 indicate that this ‘natural revovery through protection (and monitoring)’ approach has been successful, with new bed structures beginning to appear, and greater general biodiversity being observed (DAERA, personal communication).

When restoring *M. modiolus* beds, the consideration of the following aspects has been recommended: substrate, abundance of adult individuals, hydrodynamic regime and larval dispersal, natural recruitment and commercial fisheries management (for all benthic species) in the vicinity of the restoration site (Roberts *et al.,* 2011). Viability of such projects would also need to be considered in light of the species’ temperature preferences and climate change.

Costs of *M. modiolus* restoration

Overall or unit costs for horse mussel restoration projects could not be determined / located. The Strangford Lough project outlined above only reports on hatchery costs (in excess of £8,100 per month (2020 prices)) (Roberts *et al.,* 2011).

3.4.5. Honeycomb worm (*Sabellaria alveolata*) reef restoration

To date there have been no published substantial restoration initiatives for *S. alveolata* (Reach *et al.*, 2015). The company Tidal Lagoon Swansea Bay supported an academic MSc thesis project by Swansea University in 2014, which trialled the translocation of boulders covered with *S. alveolata* from a donor to a receptor site. The results of the initial pilot study showed that, in general, the translocated *S. alveolata* survived at the receptor site and reefs even appeared more vigorous five weeks post translocation. Given that this pilot study was a 6-month student project, there is no information about the sustainability and longevity of the intervention. While the project highlighted the possibility of direct translocation of *S. alveolata*, any considerations and discussions of restoring reefs are at an early stage. Value for money needs to be considered, since direct translocation of substantial reefs is likely to be costly, with a high risk of failure.

Since, as noted in Section 3.3 above, artificial structures and materials seem to be readily colonised by *S. alveolata* when located near existing reefs, creating new reefs by providing suitable settlement substratum in close proximity to existing reefs is likely to be feasible (provided environmental conditions, including hydrodynamics and food supply are also suitable). This approach, together with other concepts for the restoration of such reefs, would likely need to be subject to further research before they could be recommended to those wanting to undertake restoration.

Where there has been relatively small scale, one-off, damage at *S. alveolata* reefs, there is evidence to suggest that such areas of limited damage could naturally repair themselves rapidly (within weeks). This can occur through both the tube-building activities of adult worms, and also planktonic larval recruitment from nearby undamaged reef areas (Vorberg, 2000; Cunningham *et al.*, 1984). Vorberg (2000) found the daily growth rate of adult worms after a period of limited damage was significantly higher than undisturbed growth. It was however considered likely that *S. alveolata* reefs will not recover in such fashion if damage or disturbance were to occur relatively frequent, e.g. in areas subject to fisheries trawling.

Costs of S*. alveolata* restoration

Overall or unit costs for *S. alveolata* restoration projects could not be determined / located.

3.5 Natural Accounting and Ecosystem Services

3.5.1 Background / definitions

In order to inform any marine habitat restoration work, it is important to understand the value and services of the habitats which are being restored or enhanced as clearly as is possible.

In this context, natural capital is an often utilised concept; this can be defined as 'the world's stocks of natural assets which include geology, soil, air, water and all living things'. It is from this natural capital that humans derive a wide range of services, often called ecosystem services, which make human life possible (World Forum on Natural Capital, 2019). Thus, the marine habitats and reefs which are the subject of this report with their flora and fauna, make up part of the natural capital of Welsh marine waters.

Ecosystem services can be defined as 'the outcomes from ecosystems that directly lead to good(s) that are valued by people' (Austen *et al.*, 2010). The ecosystem services framework explicitly links ecosystem structure, processes and functioning to outcomes in the form of services which contribute to human wellbeing / welfare. Intertidal habitats in particular have long been known to be very valuable habitats which provide a wider range of beneficial ecosystem services. The evidence regarding the key ecosystem services that the six habitats which are the focus of this project deliver is summarised in the following sections.

3.5.2 Key Ecosystem Services of Focus Habitats

Saltmarsh and mudflats

Saltmarshes and mudflats provide a range of ecosystem services, including nutrient cycling, habitat provision, carbon sequestration, coastal protection, wastewater purification / detoxification, research and tourism.

Saltmarshes are generally considered to be one of the most productive ecosystems in the world, rivalling that of intensive agriculture (Niering and Warren 1980; Peterson *et al.*, 2008) and fulfil important functions in providing other marine habitats (and their fauna) with nutrients and fixed carbon (McKinney *et al.*, 2009). Intertidal mudflats are also important in the functioning of estuarine systems and may have a disproportionately high productivity compared to subtidal areas (OSPAR, 2009)). In turn, this highly productive ecosystem supports macroinvertebrates (secondary production) and provides an important year-round feeding ground, for example, for fish and wading birds (including birds such as Dunlin and Curlew, designated features of many Welsh MPAs, including the Severn Estuary SPA).

Many juvenile fish, crustaceans and molluscs use saltmarshes as nurseries. When vascular plants die, the plant matter is broken down by microbes, invertebrate detritivores, deposit and filter feeders, which are in turn predated upon by bivalves, shrimp and fish (Pennings and Bertness, 2001). Intertidal mudflats have a low species diversity but very high overall invertebrate productivity, resulting in an important and perpetually exploited food source for fish (and birds) (OSPAR, 2009). Juvenile stages of many fish species (including several commercial species) feed and find refuge amongst saltmarsh vegetation and within its shallow creeks (Dickie *et al.*, 2014). For example, Laffaille *et al.* (2000) showed that saltmarshes play a fundamental role in the feeding of juvenile sea bass, which ingested great quantities of live and dentritic organic matter, even though foraging in the vegetated areas was only possible for about 5% of the tides.

The most notable fish predators on intertidal mudflats are sole, dab, flounder and plaice which feed on polychaetes, young bivalves and other molluscs (Jones *et al.*, 2000). Mudflats are thought to be at least twice as productive as their subtidal counterparts (Elliott and Taylor, 1989). Moreover, saltmarshes and mudflats provide breeding grounds and resting / roosting areas for birds such as Lapwing, Redshank and many species of gulls and waders. At low tide, mudflats provide feeding and resting areas for internationally important populations of migrant and wintering waterfowl, whereas at high tide they are also important nursery areas for flatfish and feeding grounds for numerous fish species (OSPAR, 2009).

The slowing of wave action and water currents by saltmarsh plants helps to shelter coasts from erosion (Pennings and Bertness, 2001). Saltmarsh can significantly increase attenuation of incident waves compared to unvegetated sand / mudflats, which is especially relevant with the increased risk of sea level rise and an increase in storm frequency (Möller, 2006; Möller *et al.*, 2014). Filamentous algae, cyanobacteria and macrophyte roots strengthen sediment, further supporting erosion control (Aspden *et al.*, 2004). Saltmarshes accumulate sediment and organic matter at a rate that tends to compensate for sea level rise (Morris and Gibson, 2007). Mudflats also help protect coastal margins from erosion by dissipating wave and current energy (Bale *et al.*, 2007).

In areas receiving pollution, saltmarsh sediments sequester contaminants such as mercury, heavy metals (OSPAR, 2009; Coehlo *et al.*, 2009) and other substances such as uranium (Church, 1996). Saltmarsh plants have been shown to lead to TBT remediation in sediments (Carvalho *et al.*, 2010), and are able to regulate faecal pollution (Kay *et al.*, 2005). Microbial saltmarsh assemblages carry out nitrogen (N) and carbon (C) fixation services (Aspden *et al.*, 2004). Benthic microalgae on mudflats play significant roles in biogeochemical reactivity (MacIntyre *et al.*, 1996). With regard to water quality and nutrient cycling, coastal saltmarsh vegetation is involved in the regulation of water purity through the take up of excess inorganic nutrients such as nitrates and phosphates, therefore reducing the potential for eutrophication (Peterson *et al.*, 2008). The vegetation found on saltmarshes is also an important nutrient sink through the generation of plant biomass (Verhoeven *et al.*, 2006).

While the extent of saltmarsh habitats on a global scale might be relatively small (<2% of the ocean's surface), they are 'hot spots' for carbon burial and have a significant role to play in global carbon storage (Duarte *et al.*, 2005, Laffoley and Grimsditch, 2009, Chmura *et al.*, 2011). Saltmarsh plants are thought to have the highest carbon burial rate per unit area compared to other blue carbon habitats (Stewart and Williams, 2019), with total global sequestration rates of between 5 and 87 Mt C yr^{-1} (Chmura *et al.*, 2003) and 10.2 Mt C yr^{-1} (Ouyang and Lee, 2014) quoted in the literature.

Saltmarshes do this by slowing the rate of water flow through their roots and capturing CO_2 from the surrounding air and water column, subsequently storing it in their roots and rhizomes before exuding it into the soil. The roots themselves also physically bind together particles within the soil and encourage rhizomal microbes to do the same, trapping organic material (Ford *et al.*, 2016) and subsequently creating an anaerobic, carbon-rich sediment (Reid and Goss, 1981; cited in Ford *et al.,* 2016). The anaerobic nature of the soils (resulting in slow decomposition) means carbon can be accumulated without the soil ever reaching saturation; potentially storing carbon over millennial timescales (Stewart and Williams, 2019).

Large amounts of carbon have been calculated to have already been buried / sequestered in saltmarsh sediments globally, with levels as high as 430 Mt quoted by Chmura *et al.* (2003) for the upper 50 cm of tidal saltmarsh sediments. Sequestration rates in UK saltmarsh range from 64 to 219 g C m^{-2} yr^{-1} (Adams *et al.*, 2012), with typical figures around 120 to150 g C m^{-2} yr^{-1} (Beaumont *et al.*, 2014). A 2015 Welsh study reported on by Ford *et al.* (2019) sampled a total of 23 saltmarsh sites to determine carbon stocks. Plant and soil characteristics were analysed for each site, and the carbon stock determined for each of the sampling locations (51 in total across the 23 sites). Stored carbon calculated for the top 10 cm of soil varied from 32 t C ha^{-1} (or 3.2 kg C m^{-2}) for the *Atriplex portulacoides* vegetation class to 50 t C ha^{-1} for the *Juncus gerardii* vegetation class. Sandy soils were found to store less carbon (average 29 t C ha^{-1}) than non-sandy soils (43 t C ha^{-1}).

While not as proficient in sequestering carbon as saltmarshes, mudflats can store and sequester carbon in both organic and inorganic (carbonate) forms; ample supply permitting. For example, Sanders *et al.* (2010) found intertidal mudflats in the vicinity of mangrove forests to be sites of large organic carbon accumulation; storing almost four times the global average for sequestration in mangrove forests and suggesting that large fluxes of organic carbon produced and sequestered in mangrove forests are deposited and stored in mangrove margins and intertidal mudflats (Sanders *et al.,* 2010). Similarly, Cook (2002) found organic matter present in estuarine mudflats in Tasmania did not originate within the mudflats, instead having predominantly terrestrial sources, such as near shore estuarine transport (driven by riverine input) as well as direct terrestrial run-off and reworking of glacial and post-glacial sediments. The literature therefore suggest that mudflats in the vicinity of other blue carbon habitats and / or nutrient sources may facilitate the storage and sequestration of excess carbon.

A 2020 report on the carbon sink potential of the Welsh marine environment found that, by area, saltmarshes and intertidal flats sequester the most carbon in the Welsh marine environment, at estimated rates of just under 15,000 tonnes of carbon every year (NRW, 2020).

Seagrass beds

Seagrass beds provide a wide range of ecosystem services, including raw materials and food, coastal protection, water purification, maintenance of fisheries, tourism and recreation.

Seagrass leaves slow currents and water flow rates, and their roots and rhizonmes stabilise the sediments (van der Heide *et al.,* 2011; Hansen and Reidenbach, 2012). The settlement and accumulation of fine sediments, decaying detritus and larvae (Turner and Kendal, 1999) into the soils is facilitated, thus increasing nitrogen burial and protecting against wave disturbance (Middelburg *et al.,* 2004; Barbier, 2011). Remineralisation of nutrients and nitrogen fixation then occurs within the sediment, fuelled by seagrass photosynthate (oxygen and carbon) released through the roots. This stimulates coupled nitrification–denitrification, leading to permanent N removal (Iizumi *et al.,* 1980; Aoki and McGlathery, 2018), thus increasing coastal water quality..

Seagrass beds create a three-dimensional structure in what would otherwise be a far less complex seabed habitat. They are furthermore highly productive and provide shelter, protection, hiding places and substrata for many other species, thus promoting diversity and species richness (Davison and Hughes, 1998).

Many commercially important fish species utilise seagrass beds as spawning, nursery and feeding habitat (Barbier, 2011; Boudouresque *et al.,* 2012; Green and Short, 2003; Pergent *et al.*, 2012). Seagrass sediments also supports a rich infauna of polychaetes, bivalve molluscs and burrowing anemones and are an important food source for waterfowl.

Seagrass beds can attract recreational activities such as snorkelling and diving, due to increased species diversity, abundance and water clarity. Substantial seagrass beds are also known to provide a coastal protection function; as dense beds slow water movement and reduce wave action. Moreover, seagrass fronds are a well-utilised raw material in some areas of the world, and can be used in roof insulation, weaving, animal feed and compost. Their sensitivity to nutrient enrichment has also led to the employment of seagrass beds as bioindicators; such that the dying-off of large areas of the bed may indicate heavy-metal contamination or eutrophication, and as such they can be used to prevent a potentially large-scale pollution event.

The ability of seagrass foliage to sequester CO_2 dissolved in seawater is also an important ecosystem service; with organic carbon in seagrass sediments thought to be stored over decadal to even millennial time scales (Kennedy *et al.,* 2010; cited in Greiner *et al.,* 2013).

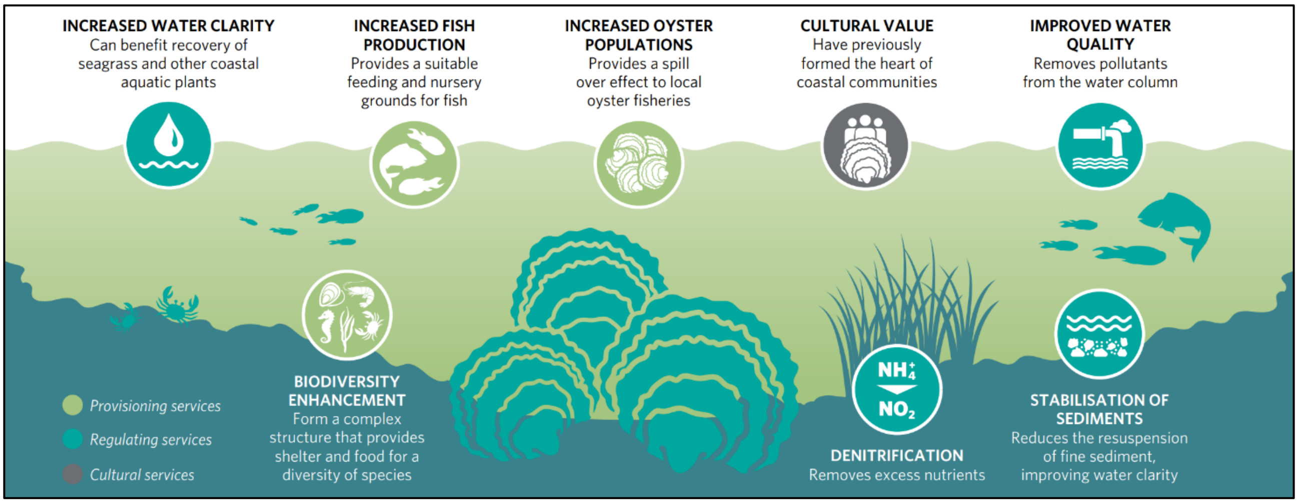
A mean net sequestration rate of 83 g C m^{-2} yr^{-1} and a total global storage of 19.9 Pg C (billion tonnes carbon) has been estimated within the top 100 cm of the world’s seagrass sediments (Green *et al.,* 2018). As such, it has been suggested that, although these plants only cover a relatively small area of the global ocean floor (0.1-0.2%), they are responsible for between 10 and 18% of the total carbon storage in the ocean (Laffoley and Grimsditch, 2009; Green *et al.,* 2018).

A recent UK study compared carbon content in sediment cores taken from the upper 30 cm and 100 cm in subtidal seagrass sediments of 13 seagrass meadows in south-west England (Green *et al.,* 2018). The authors found that the 100 cm samples contained a carbon store three times higher than samples taken solely from the top 30 cm; 41.54 ± 4.54 Mg C ha^{-1} (30 cm depth), 140.98 ± 73.32 Mg C ha^{-1} (100 cm depth). When converted to carbon stored to a depth of 25 cm, Green *et al.* (2018) determined that the studied (English) seagrass meadows fell within the upper range of those recorded in the rest of Europe. The authors state that 'across Europe, estimates of *Z. marina* carbon stock vary considerably, ranging from 500 ± 50.00 g C m^{-2} to 4,324.50 ± 1,188.00 g C m^{-2} in the top 25 cm of sediment. With an average carbon stock of 3,372.47 ± 1,625.79 g C m^{-2}, the UK is second only to Denmark'.

The 2020 NRW carbon sink report noted that seagrass beds, on a per-unit area basis, sequester almost as much carbon as saltmarshes, although, due to their relatively small total area in Wales, overall sequestration rates are fairly low at just under 200 tonnes of carbon per annum.

Native oyster habitat

Native Oyster habitats provide various ecosystem services, including improvement of water quality, removal of excess nutrients / contaminants, habitat provision, recreation, carbon sequestration and the supply of a commercially important food source. Key services have been summarised in an infographic by Preston *et al.* (2020), see Image 12.



Source: Preston *et al.* (2020)

Image 12. Native Oyster ecosystem services

Oyster habitats reduce the resuspension of fine sediment in the water column through the filtering of algae (suspension-feeding) and suspended particulates (including organic matter i.e. nutrient enrichment from sewerage plants / agricultural run-off), producing faeces and pseudofaeces that are then deposited into the sediment. The rate of biodeposition achieved by living native oysters is thought to be three times that of control conditions and conditions with dead oysters (Lee *et al.,* 2020). These processes can significantly improve surrounding water quality and prevent eutrophication. This is important for other habitats and species in the surrounding water body, especially in areas prone to nutrient enrichment, pelagic fisheries and / or areas of commercially important fish spawning. Moreover, the decrease in turbidity improves water clarity, which can benefit the recovery of seagrass beds and other coastal aquatic plants, increasing biodiversity and attracting recreational water users (such as divers), thus also providing socio-economic gain.

Dense beds of these suspension feeding bivalves are furthermore important for nutrient cycling in estuarine and coastal ecosystems (known as pelagic-benthic coupling), as the production of faeces and pseudofaeces in these habitats enriches the underlying sediment, providing a rich food source for infaunal detritivores, deposit feeders, meiofauna and bacteria (Dame, 1996).

Native Oyster habitats are a known nursery and feeding habitat for spawning fish, as the 3D structure of the reef provides a refuge from predation and the high biodiversity found in reefs supplies a varied food source for young fish. Furthermore, native oysters themselves are a commercially important species, having been cultivated and fished around the coastline of the UK for millennia. So much so, that in 1864, 700 million oysters are thought to have been consumed in London alone (Preston *et al.,* 2020).

Native Oysters, in common with all other shellfish species, assimilate carbon in the form of calcium carbonate, via shell production, with carbon comprising (on average) 11.7% of shell material (van der Schatte *et al.*, 2020). During the calcification process, CO_2 is formed; potentially leading its release into the atmosphere. As such, shellfish bed habitats, are often considered to be a source of atmospheric CO_2 (Fodrie *et al.*, 2017). However, shellfish beds also induce passive sedimentation of particles from the water column, potentially trebling carbon downdraw through biodeposition alone (see Lee *et al.,* 2020). Oyster habitats furthermore produce and deposit of faeces / pseudofaeces containing particulate organic matter into the underlying sediment, where it can be stored (Preston *et al.,* 2020). Most of the studies on carbon sequestration / storage potential of oysters have focussed on American species. Fodrie *et al.* (2017) sampled 22 eastern oyster reefs (*Crassostrea virginica*) in Northern Carolina, United States, and found that some of the reefs had functioned as net CO_2 sinks, namely those fringing saltmarshes and those located in the shallow subtidal. A study by Higgins *et al.* (2011) estimated that one (American / Chesapeake Bay) oyster reef could remove a total of 13.47 ± 1.00 t C ha^{-1} yr^{-1} in a single growing season at a density of 286 oysters m^{-2}. These studies therefore suggest that carbon burial and sequestration is likely dependent on the density of oyster habitats and underlying sediment type. As UK oyster habitats tend to have much lower densities, carbon sequestration and storage is likely to be much lower.

Horse mussel beds

M. modiolus beds provide several ecosystem services, such as nutrient cycling and deposition, habitat provision and carbon storage.

Horse mussel beds have been observed to enhance sedimentation by 30% and contribute to the downward flux of material to the seabed through active filter feeding and the reduction of flow rate, doubly enhancing deposition (Kent *et al.,* 2017) of inorganic nutrients into the sediment. This behaviour has important benefits to the wider ecosystem through processes such as benthic-pelagic coupling, sediment stabilisation and water purification, which provide a stable, nutrient-rich environment for benthic infauna and increases species richness and diversity as a result. Similarly, the burrowing behaviour of bivalves and the production of pseudofaeces can also influence sediment resuspension, which is important for resuspending nutrients, boosting feeding activity and enhancing primary productivity in the upper water column (Kent *et al.,* 2017).

M. modiolus beds also provide a ‘habitat provision’ ecosystem service (Kent *et al.,* 2016), whereby the complex 3D nature of horse mussels beds facilitates an important feeding and nursery habitat for commercially important species such as common whelks, queen scallops and spider crabs. These habitats also supply protection from predators and attract prey species to sustain a healthy population. Whilst horse mussels are not a commercially important bivalve species themselves, the presence of their beds increases the abundance of commercially available species and thus augments catch rates on the reef when compared to off-reef habitats (Kent *et al.*, 2016). In terms of monetary value, the common whelk is among the top three most valuable shellfish species caught in Wales, with landings of 5,000 tonnes in 2013, worth £3.6 million (MMO, 2014).

Like native oysters, horse mussels additionally provide carbon storage as an ecosystem service. The accumulation of biogenic carbonate in mussel shells, high sedimentation rates and biodeposition behaviour is likely responsible for the burial of carbon within mussel bed sediments. The creation of this ‘mussel mud’, comprised of pseudofacaes, faeces, sediment and dead shells has the potential to store carbon for over 1,000 years (Mainwaring *et al.*, 2014). For example in the Firth of Lorn, *M. modiolus* accounted for 94% of carbonate standing stock in the mussel bed community, but only 38% of the estimated carbonate production (Collins, 1986). Furthermore, their carbonate is considered to degrade very slowly, given their large, robust shells; with bioerosion on temperate shelves thought to require a timescale of centuries to several millennia for total shell destruction (Smith and Nelson, 2003).

Sabellaria alveolata reefs

Sabellaria alveolata reefs are thought to mainly provide ecosystem services in the form of habitat provision, nutrient regulation and coastal protection.

S. alveolata reefs have been shown to increase the abundance and species richness of benthic fauna (Dubois *et al.*, 2002) which, in addition to the reef-building worms themselves, are utilised as a source of food by some demersal fish species. The reefs provide additional attachment surfaces, crevices and spatial heterogeneity which are otherwise rare in these habitats they reside on (Dubois *et al.,* 2002; Pearce *et al.*, 2011). The density of living worms within a reef provides an indication of its health in terms of its reproductive success, and also gives a strong indication of the health of the associated benthic communities; with more diverse and abundant benthic communities being associated with the healthiest reefs in terms of living *S. alveolata* density.

It is also possible that reef-building worms provide some biological enhancement by sequestering nutrients from the water column and making them available for other marine life (Pearce *et al.*, 2011). These often substantial structures are also thought to provide a coastal protection function, slowing subtidal water movement and reducing current strength upon arrival at the shore (Bonifazi *et al.,* 2019).

Habitats formed by reef-building polychaetes *Sabellaria alveolata* and *S. spinulosa* consist of agglutinated sand grains and shell fragments (Naylor and Viles, 2000). In this respect they differ from the calcium carbonate tubes of worms such as *Serpula vermicularis*, which are secreted *de novo* by the worms. Sabellaria reefs are therefore merely a temporary structural rearrangement of sand and shelly particles from the surrounding sediment, and as such likely have very limited carbon sequestration and storage potential.

3.5.3 Summary

The studied habitats have the potential to contribute to multiple ecosystem services; the extent to which this happens will depend on various factors, including location and habitat quality. In some cases, the evidence review presented above did not identify a link between a habitat and an ecosystem service; this does not necessarily mean that it makes no contribution, simply that the evidence base is lacking.

Tables 7 to 9 below present a summary of the ecosystem services provided by the coastal and marine habitats studied for this project. The potential contribution of a given habitat to an ecosystem service is indicated, from high (H) to low (L); 'A' denotes cases in which beneficial ecosystem services have been assumed to exist, but where direct evidence is currently lacking. The ecosystem service framework applied for this table is based on the latest abridged version of the Common International Classification of Ecosystem Services (CICES) (European Environment Agency, 2018).

Ecosystem services associated with marine and coastal habitats studied for this report

Key for tables 7, 8 and 9

H: High potential contribution of habitat to ecosystem service	M: Medium potential contribution of habitat to ecosystem service	L: Low potential contribution of habitat to ecosystem service	A: Assumed beneficial service (though no direct evidence for this habitat)	N: No evidence/no link

Table 7. Provisioning ecosystem services associated with marine and coastal habitats studied for this report

CICES ecosystem service category	Saltmarshes contribution to ecosystem service	Mudflats contribution to ecosystem service	Seagrass beds contribution to ecosystem service	*Sabellaria alveolata* contribution to ecosystem service	Oyster habitats contribution to ecosystem service	Horse mussel beds contribution to ecosystem service
Marine plants for nutrition, materials or energy	L	L	N	N	N	N
Marine animals for nutrition, materials or energy	M	M	M	N	H	N

Table 8. Regulating ecosystem services associated with marine and coastal habitats studied for this report

CICES ecosystem service category	**Salt-marshes contribution to ecosystem service**	**Mudflats contribution to ecosystem service**	**Seagrass beds contribution to ecosystem service**	***Sabellaria alveolata* contribution to ecosystem service**	**Oyster habitats contribution to ecosystem service**	**Horse mussel beds contribution to ecosystem service**
Bioremediation by micro-organism, algae, plants and animals	H	M	M	L	H	H
Filtration/sequestration/storage/accumulation of carbon	H	M	H	N	L	M
Control of erosion	M	M	M	A	L	N
Buffering and attenuation of mass movement	H	M	M	A	A	N
Hydrological cycle and water flow regulation (including flood control, coastal protection)	H	M	M	A	M	N
Maintenance of nursery populations and habitats	H	M	M	M	M	M
Maintenance of habitat for charismatic species	H	H	H	N	N	N
Regulation of the chemical condition of salt waters by living processes	N	L	L	N	H	A

Table 9. Cultural ecosystem services associated with marine and coastal habitats studied for this report

CICES ecosystem service category	Saltmarshes contribution to ecosystem service	Mudflats contribution to ecosystem service	Seagrass beds contribution to ecosystem service	Sabellaria alveolata contribution to ecosystem service	Oyster habitats contribution to ecosystem service	Horse mussel beds contribution to ecosystem service
Characteristics of living systems that enable recreation (passive and active)	H	L	M	A	L	A
Characteristics of living systems that enable knowledge generation (scientific, traditional)	H	M	M	M	M	A
Characteristics of living systems that enable education and training	L	L	L	A	A	A
Characteristics of living systems that enable creative activities	L	L	M	A	A	A
Characteristics of living systems that have symbolic, sacred and or religious meaning	L	L	M	N	N	N

4. The Opportunity Datalayers

4.1 Introduction

This section provides background information on the opportunity datalayers which were created for this project, as well as those which have been produced by JNCC. It is structured according to the habitats which formed the focus of the datalayers, namely:

- Mudflat and saltmarsh (Section 4.2);
- Seagrass (Section 4.3);
- Native oyster data (Section 4.4);
- Horse mussel (Section 4.5); and
- Honeycomb worm.

Please note that maps for other (non-opportunity and input) datalayers which have been created as part of this project are provided in Appendix A.

4.2 Mudflat and Saltmarsh

One datalayer was produced in relation to mudflat and saltmarsh habitats, as noted in Section 2.2. This polygon datalayer was created to show where intertidal habitats (mainly mudflats and saltmarshes) could potentially be restored, or created, in the current floodplain, by highlighting areas behind shoreline stretches where 'Managed Realignment' or 'No Active Intervention' has been selected as a potential approach / policy as part of the Shoreline Management Plan (SMP) process.

SMPs are high-level plans that set out where the coastline should be defended, and where it would be more sustainable to adapt over time. They break the coastline down into smaller sections known as 'policy units', and explain how the policy units should be managed over three 'epochs', the short-term (2005-2025), the medium-term (2025-2055) and the long-term (2055-2105). One of the following management policies is assigned for each period of time:

(1) No active intervention, where there is no investment in coastal defences and natural processes are allowed;
(2) Hold the (existing defence) line, by maintaining or changing the existing standard of protection;
(3) Managed realignment; and
(4) Advance the line, by extending new defences in to the sea (none applied in Wales).

It is important to note that SMPs are non-statutory; thus, even when an SMP says 'hold the line' or 'managed realignment' for example, funding often still has to be secured to deliver the policy intent (NRW, 2021c).

The datalayer was created by combining / overlaying two existing layers, as noted in Table 5 in Section 2.2. These are: the Welsh / NRW Zone 3 Floodmap, and the Welsh / NRW

SMP layer. The former shows land assessed as having a 1 in 100 or greater annual probability of river flooding (>1%), or a 1 in 200 or greater annual probability of flooding from the sea (>0.5%) in any year'. It can act as a good proxy for land potentially suitable for managed realignment, as it provides a good indication of suitable elevation in the tidal frame. Depending on location, some percentage of a floodplain would be expected to be above the levels where intertidal habitats establish (around HAT, see Section 3.3.1); however, this may change in the future, given anticipated rates of accelerated sea level rise. The SMP layer highlights what type of policy (see previous paragraph) has been selected for a given stretch of shoreline and epoch. In combining / overlapping the datalayers, floodplains were effectively categorised according to SMP policy for each epoch. A figure showing the extent of the datalayer, as well the the various policies which would apply to the floodplains across the different epochs, is shown in Figure 1.

This datalayer should very much be seen as an aide to initiating a search for a potential site, rather than the sole tool for identifying a suitable site. Several key limitations are worth highlighting.

- No land use filtering has been undertaken, and the datalayer thus contains many areas which would typically be considered unsuitable for managed realignment, notably urban areas, smaller settlements, industrial units, landfill sites, etc.
- Neither international nor national conservation designations were used to exclude sites, however, avoiding sites which are thus designated is often practiced when undertaking a site search for managed realignment schemes, as this avoids the potential need for compensation in this regard.
- Areas being subject to an SMP policy other than managed realignment does not preclude them from managed realignment being undertaken in them.
- The size of the Welsh floodplains ranges widely, from 10 ha to over 10,000 ha. Such large sites are unlikely to ever be required, nor would it be likely that undertaking managed realignment or RTE across the whole extent of such a large site would be feasible. For example, in certain areas of a given estuary (particularly in narrower confined upper reaches), undertaking too big a scheme would likely be unduly detrimental to the adjacent system, and thus be undesirable. Thus, in reality, large sites could be sub-divided, and those areas which are lowest lying and closer to the shoreline preferentially targeted; and
- Even apparently ideal sites with regard to elevation and landuse may not be suitable or available for intertidal habitat creation.

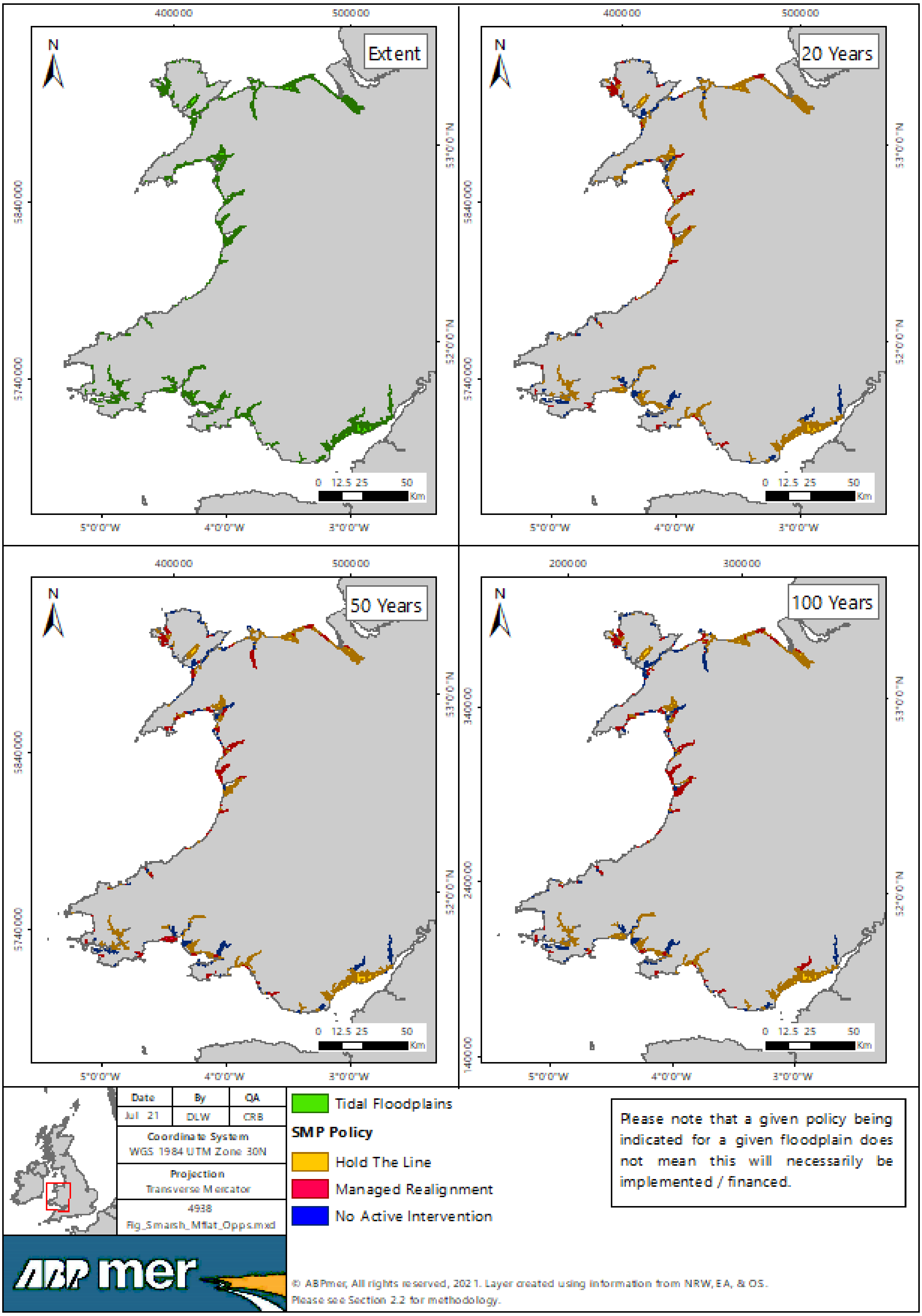


Figure 1. Welsh tidal floodplains and applicable SMP policies

It is beyond the scope of this project to provide detailed advice on how to undertake a site selection exercise and initial feasibility review for managed realignment and RTE sites. Such guidance can, for example, be found in Nottage and Robertson (2005) and Leggett *et al.* (2004), as well as the upcoming saltmarsh guide noted in Section 3.4.1. However, in summary, the following broad steps may be required when undertaking a search for a potential site, and highlight how the datalayer produced for this project could fit into such a search process. The order, and need for, some of the steps would largely depend on the motivation for a given search.

- Identify sites broadly suitable for managed realignment or RTE in the region of interest (using the datalayer created for this project; excluding areas with unsuitable landuse);
- Create a list of most desirable sites (e.g. in terms of vicinity to damaging development);
- Obtain LiDAR data to create a digital elevation model for some, or all of, the selected sites, and relate the elevation to tidal levels to gain a high level understanding of potential habitats (see Section 3.3.1);
- Refine site boundaries to reflect field / ownership boundaries;
- Undertake a site characterisation process for short listed sites, investigating constraints, opportunities and potential habitats, by researching aspects such as: existing nature conservation designations, shoreline management policy, cultural heritage records, contamination, land use and infrastructure, drainage outfalls, need for landward defence, likely impacts on fronting system (e.g. by calculating a high level tidal prism value), etc. (consult local knowledge if possible);
- Rank the sites according to suitability and project specific drivers; and
- Make enquiries with landowners (if not already the owner).

The attributes table in the datalayer produced for this project can be used to inform the initial search for a potential site by, for example, only displaying sites in a given estuary, only selecting areas where the short and medium-term SMP policy is managed realignment, etc..

Once a site, or a set of sites have been identified as the preferred site(s), further studies would generally be required; these would typically involve the commissioning of specialist consultancies, and may include the development and assessment (including numerical modelling) of preliminary designs. Early stakeholder and regulator engagement is also recommended. Climate change implications may furthermore need to be studied, for example, by considering the sustainability of habitats in light of projected rates of sea level rise, and other aspects such as storminess.

4.3 Seagrass

JNCC is currently in the process of finalising a UK-wide datalayer which will indicate where opportunities for the restoration / creation of *Z. marina* beds may exist. In order to create this layer, a suite of environmental variables were selected as model inputs, based on their ability to influence the growth and survivability of *Z. marina* habitats. The JNCC model was furthermore restricted to a depth range of between 0 and 15 m. The resulting datalayer depicts probability of occurrence on a scale of 0 to 1; the higher the values, the more likely it is that a given area will be suitable. A draft map depicting this UK-wide layer is provided in Figure 2 (from Castle *et al.*, in prep).

Source: Castle *et al.*, in prep.

Figure 2. JNCC draft map showing mean predictive values of habitat suitability for *Zostera marina* beds across the UK, within the model extent of 0 to 15 m depth

For detailed advice on how this layer should be utilised, as well as its limitations, related JNCC reporting, once available, should be consulted.

This layer can be used as an aide when undertaking a site selection exercise for seagrass bed restoration sites. Whilst it is beyond the scope of this report to provide detailed guidance, it is noted that such guidance is currently being produced, as previously mentioned in Section 3.4.2 above. In summary, the following broad steps may be required when undertaking a search for a potential site, and highlight how the datalayer produced by JNCC could fit into such a search process. The order, and need for, some of the steps would largely depend on the motivation for a given search.

- Identify areas where there are existing and historic sites in the areas of high probability highlighted by the JNCC opportunities layer, consulting local knowledge and relevant survey data / reporting);
- Consult local knowledge to help distinguish historic from existing sites, and create a list of most desirable sites (e.g. sites where there are abundant historic records, and where there are still some habitats nearby; sites where water quality is very good);
- Undertake a site characterisation process for short listed sites, investigating aspects such as: turbidity, water quality, water depth, predominant sediment, proximity to existing seagrass habitats, existing nature conservation designations, levels of recreational boating / mooring, fishing, bait digging, etc. (consult local knowledge if possible);
- Rank the sites according to suitability and project specific drivers; and
- Consult the landowner (likely to be The Crown Estate, with some local exceptions).

Once a site, or a set of sites have been identified as the preferred site(s), further studies would generally be required; these would likely involve the commissioning of specialist consultancies or academic institutions, and may include the development and assessment of preliminary designs, as well as numerical modelling to determine suitability. Early stakeholder and regulator engagement is also recommended. Availability of suitable donor seeds or seedlings would also need to be investigated. Climate change implications may also need to be studied, for example, by considering the sustainability of habitats in light of projected rates of sea level rise, and other aspects such as storminess.

4.4 Native Oyster

One dedicated datalayer was produced in relation to native oyster beds (see Section 2.2). This polygon datalayer was created to show where native oyster beds could potentially be established in Welsh subtidal areas.

The datalayer was derived from seabed sediment, depth and current speed data (with a dedicated input layer produced on the latter as part of this project, see Section 2.2). It provides a national, high level, indication of where native oyster reefs could potentially be restored in Wales, based on some key environmental variables; Figure 3 below shows the resulting opportunity areas.

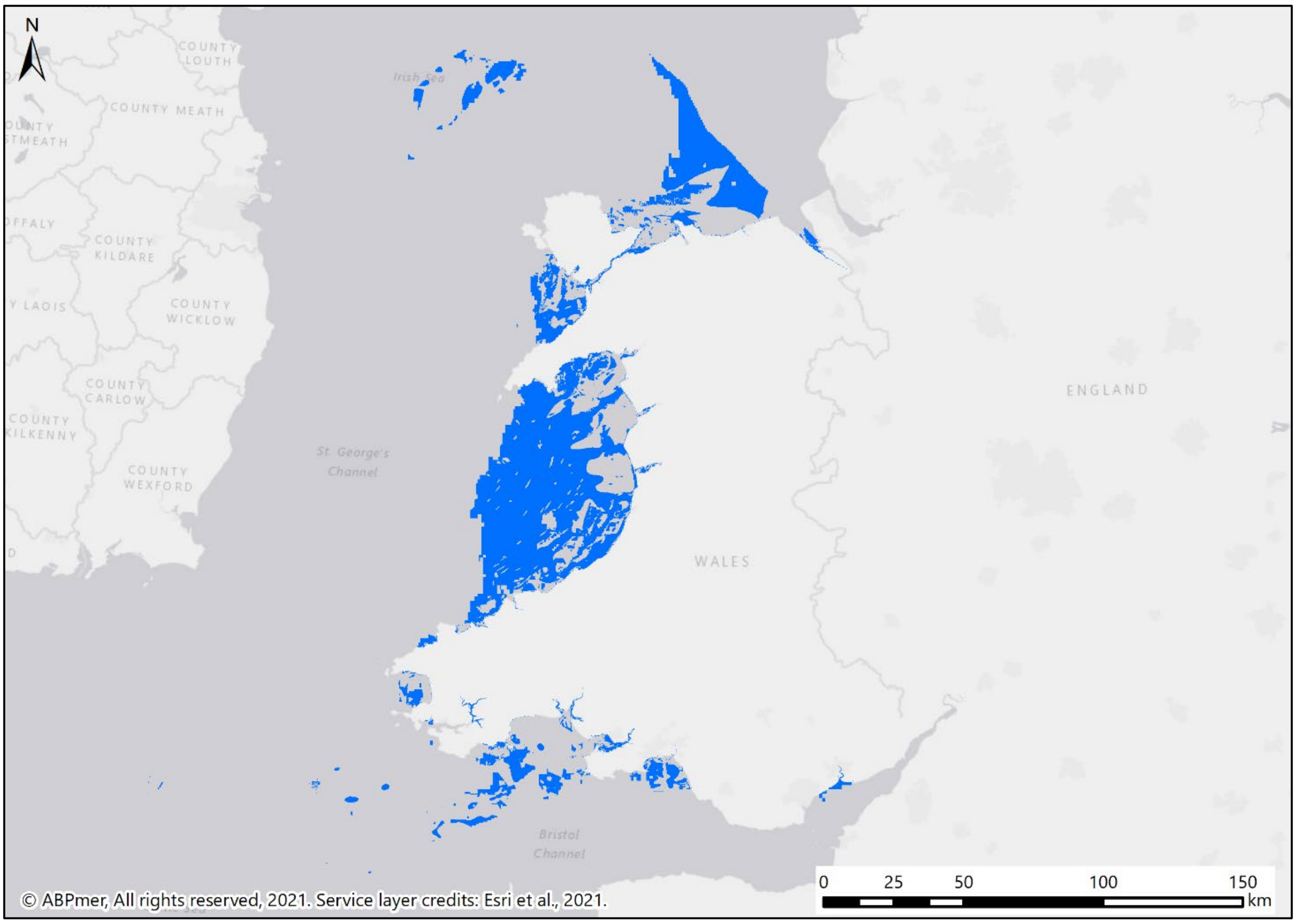


Figure 3. Native oyster opportunity areas (within the model extent of 0 to 80 m depth)

The datalayer should be considered as an initial aid to identifying sites. The following key limitations are worth highlighting:

- The areas identified are based on outputs from large scale models and should not always be assumed to be precise at the local level;
- The location of significant activities (such as dredging), marine assets (such as submarine cables) and ecological risks (such as disease control areas), which could restrict a location's potential, have not been factored in to the creation of this layer;
- It should not be assumed that all the areas identified meet all environmental conditions required to re-establish native oyster bed habitat; for example, water quality and salinity are important considerations which did not factor in the creation of this datalayer; and
- It is probable that there are some areas outside the areas identified that could be suitable for restoration.

It is beyond the scope of this project to provide detailed advice on how to undertake a site selection exercise and initial feasibility review for native oyster reef restoration. The recently produced guidance by Preston *et al.* (2020) provides useful additional detail on this topic. Broadly, based on this, and other above-noted aspects, the following steps could be followed when undertaking a regional search for a potential site for native oyster restoration (with the order, and need for some of the steps, to a large extent depending on the motivation for a given search):

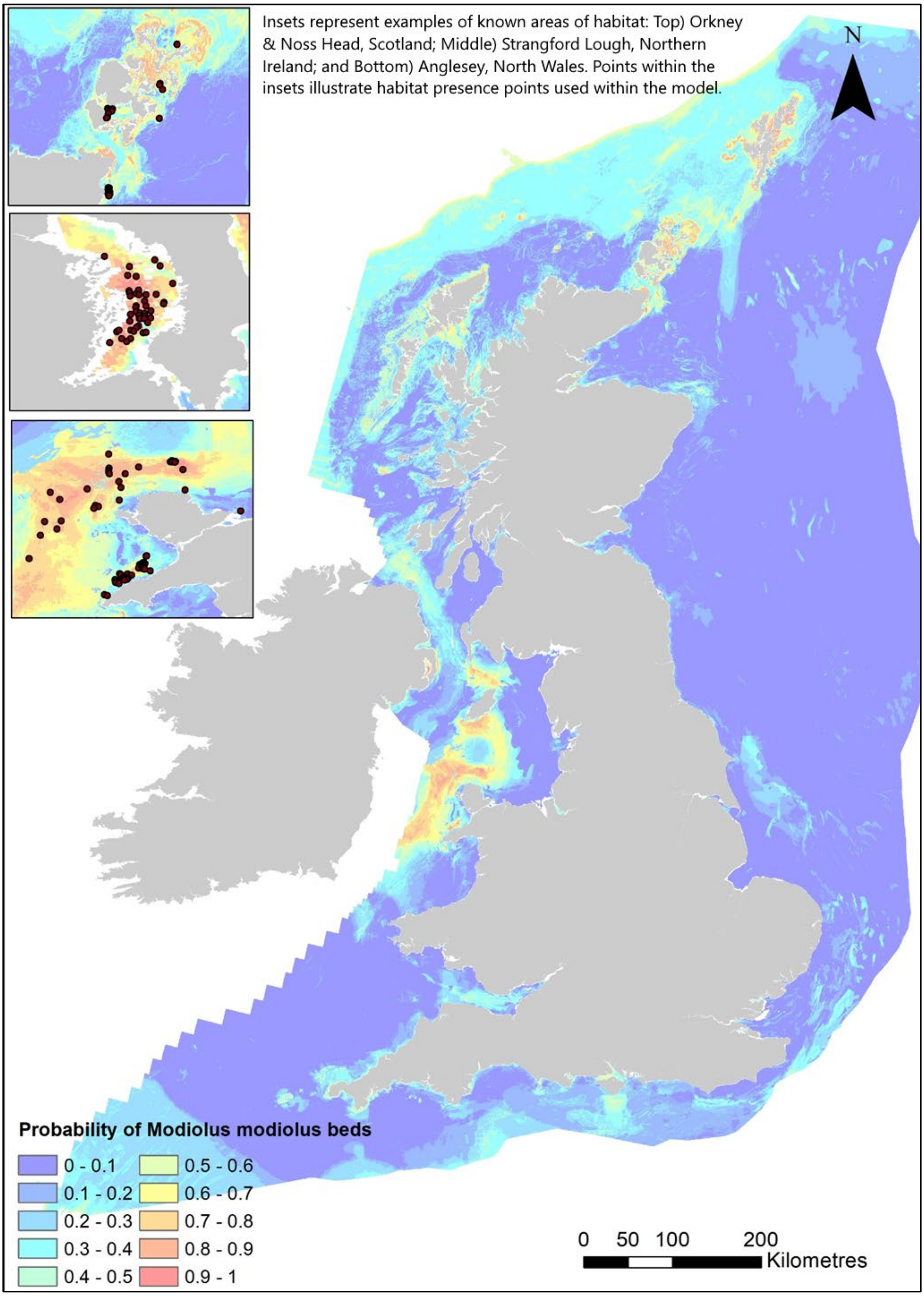
- Identify areas where there are existing and historic sites in the areas of interest highlighted by the opportunities layer, consulting:
  - Available datalayers / databases (using the ‘historic and current records’ datalayer created for this project (see Section 2.2), noting that this highlights the presence of individuals, and not reefs);
  - Local knowledge and literature;
- Undertake a site characterisation process for short listed sites, investigating aspects such as: cultch availability, salinity, turbidity, water quality, water depth, predominant sediment, proximity to existing / historic reefs, wave exposure, tidal current speeds; existing nature conservation designations; levels of recreational boating / mooring, fishing (particularly trawling, bait digging, etc.); (consult local knowledge if possible);
- Rank the sites according to suitability and project specific drivers; and
- Consult the landowner (likely to be The Crown Estate, with local exceptions).

Once a site, or a set of sites have been identified as the preferred site(s), further studies would generally be required; these would likely involve the commissioning of specialist consultancies or academic institutions, and may include the development and assessment of preliminary designs. The availability of a sufficient broodstock supply and cultch materials may also need to be investigated, as should the introduction of suitable management measures to regulate fishing activity on any newly established beds. Early stakeholder and regulator engagement is also recommended. Climate change implications may also need to be considered, for example, by considering the sustainability of habitats in light of projected rates of sea level rise, and other aspects such as storminess.

4.5 Horse Mussel

JNCC is currently in the process of finalising a UK-wide datalayer which will indicate where opportunities for the restoration / creation of horse mussel beds may exist. In order to create this layer, a suite of environmental variables were selected as model inputs, based on their ability to influence the growth and survivability of horse mussel habitats. The JNCC model was furthermore restricted to a depth range of between 0 and 300 m. The resulting datalayer depicts probability of occurrence on a scale of 0 to 1; the higher the values, the more likely it is that a given area will be suitable for restoration. A map depicting this UK-wide layer is provided in Figure 4 (from Castle *et al.*, in prep), whilst Figure 5 shows the Welsh extent of the layer.

For detailed advice on how this layer should be utilised, as well as its limitations, related JNCC reporting, once available, should be consulted.



Source: Castle *et al.*, in prep.

Figure 4. JNCC draft map showing mean predictive values of habitat suitability for *M. modiolus* beds across the UK, within the model extent of 0 to 300m depth

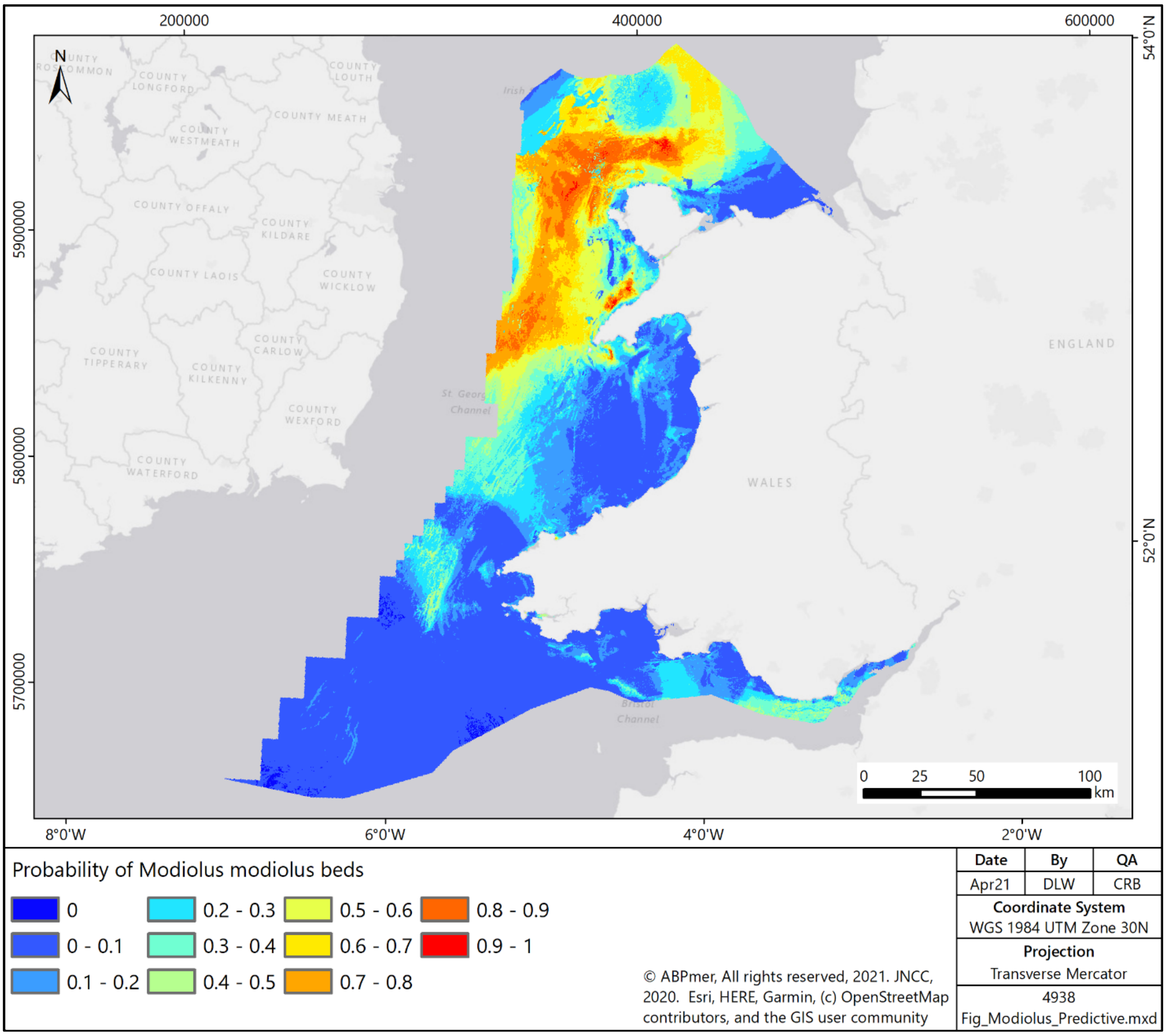


Figure 5. Map showing mean predictive values of habitat suitability for *M. modiolus* in Welsh waters, within the model extent of 0 to 300m depth

This JNCC layer can be used as an aide when undertaking a site selection exercise for horse mussel restoration sites, and it can also be used to pinpoint areas where horse mussels may already be present, where these have not previously been surveyed.

Whilst it is beyond the scope of this report to provide detailed guidance, in summary, the following broad steps may be required when undertaking a search for a potential site. The order, and need for, some of the steps would largely depend on the motivation for a given search.

- Identify areas where there are existing and historic sites in the areas of high probability highlighted by the JNCC opportunities layer, consulting:
  - Available datalayers / databases (using the 'historic and current records' datalayer created for this project (see Section 2.2), noting that this highlights the presence of individuals, and not beds);

    - Local knowledge and literature;
- Undertake a site characterisation process for short listed sites, investigating aspects such as: cultch availability, salinity, turbidity, water quality, water depth, predominant sediment, proximity to existing / historic beds, existing nature conservation designations, levels of recreational boating / mooring and fishing (particularly trawling, dredging and potting, etc.) (consult local knowledge if possible);
- Rank the sites according to suitability and project specific drivers; and
- Consult the landowner (likely to be The Crown Estate, with local exceptions).

Once a site, or a set of sites have been identified as the preferred site(s), further studies would generally be required; these would likely involve the commissioning of specialist consultancies or academic institutions, and may include the development and assessment of preliminary designs, as well as numerical modelling to determine suitability. Early stakeholder and regulator engagement is also recommended. Availability of suitable donor sites would also need to be investigated. Climate change implications may also need to be considered, for example, by considering the sustainability of habitats in light of projected temperature rises, and other aspects such as storminess.

4.6 Honeycomb Worm

As noted in Section 2.2., for *S. alveolata*, no opportunity layer has been produced. This was not considered necessary, as this species appears to spread well, chiefly through larval recruitment, where conditions are suitable (see Section 3.4.).

5. Conclusions and Recommendations

A key objective of this project has been to, where appropriate, develop and / or signpost a series of datalayers to demonstrate the spatial opportunities for the restoration of marine and coastal habitats and species in Welsh waters. The focus has been on the following six habitats / species: intertidal mudflats, coastal saltmarshes, seagrass beds, horse mussel beds, *S. alveolata* reefs and native oyster habitats. To provide the context for marine restoration and its role in increasing the resilience of marine ecosystems, a literature review has also been undertaken to document a variety of aspects, including the key legislation and drivers, mechanisms / techniques for restoration and ecosystem services provided by the habitats.

For the purpose of this project, restoration has been defined as including both the re-establishment of natural processes, ecosystem functionality and biodiversity in degraded habitats, as well as re-creating habitat where it has been lost. The relatively newly emerged term ‘nature-based solutions’ is often utilised in recent literature, and is applicable to the restoration of all of the six habitats included within this project.

The literature review has confirmed that, increasingly, there is a focus on restoration of marine species and habitats, as one approach to reversing the global decline in biodiversity. Restoring lost and degraded habitats can not only help meet legislative duties and objectives but also harness multiple benefits for the people of Wales and beyond. Many international drivers exist, and have often informed / fed into Welsh legislation and policy. In particular, the new legislative framework in Wales put in place by the Well-being of Future Generations Act 2015 and the Environment (Wales) Act 2016 formalised Welsh Government’s commitment to the sustainable management of natural resources and their drive to halt and reverse the decline in biodiversity. Notable recent international initiatives, which the UK and Wales are part of, include The Leaders’ Pledge for Nature, the Global Ocean Alliance and the upcoming 2021 United Nations Climate Change Conference, also known as COP26.

The ecosystem services review undertaken has shown that the benefits of the six habitats which have been the focus of this report are substantial; key benefits can be summarised as follows (see also summary graphic in Image 13):

- Habitat provision / biodiversity enhancement. All the habitats discussed provide important supporting habitat for a variety of flora and fauna, and offer refuges from predators and physical stress, as well as food, for a wide range of species, notably fish and birds.
- Increased fish populations. All of the habitats, albeit to varying degrees, act as feeding and nursery grounds for a large variety of fish; some of the habitats also promote the settlement of shellfish species. Many of these are of commercial value, including bass, plaice, whelk and native oysters.
- Water quality improvements, denitrification. All the habitats studied facilitate at least some improvements in water quality, through the removal of pollutants and nutrients from the water column, and also by lowering turbidity rates (particularly shellfish, by filtering algae and suspended particulates out of the water column).
- Carbon sequestration and storage (also referred to as ‘Blue Carbon’ in the context of marine habitats). Saltmarshes, mudflats and seagrasses have particular value /

potential in this regard, whilst the value of shellfish beds is somewhat diminished due to the calcification process producing CO_2 (though they may still, on balance, sequester more C than they produce). *S. alveolata* do not fulfil this service.

- Natural hazard regulation, increased resilience. Intertidal and shallow subtidal habitats attenuate waves, albeit to varying degrees, with saltmarshes likely being the most efficient. This reduces erosion at the coast, and expenditure on coastal defence infrastructure.

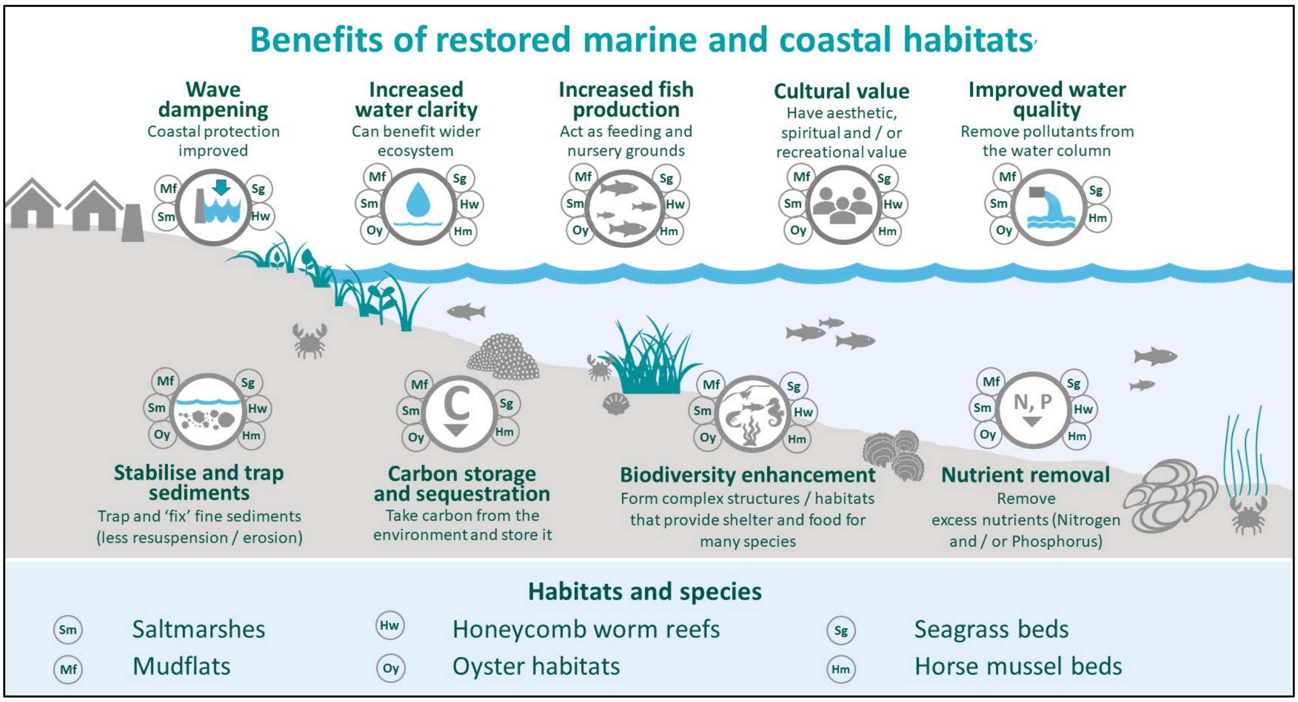


Image 13. Summary image on ecosystem service benefits of restored marine and costal habitats

All of these valuable habitats are vulnerable to the impacts of climate change, mostly through relative sea level rise, changes to wind and wave energy, temperature and precipitation. Increasing or maintaining their extent through restoration would go some way to building the resilience of the marine environment in Wales, though it is recognised that not all restoration efforts may ultimately be successful, or persist longer term. Horse mussels in particular could disappear from Welsh waters over the next century, should average sea temperatures increase significantly, as this species is already at the southern extent of its range in Wales (being a boreal species). All other species / habitats studied here are less sensitive in comparison, and would likely be able to tolerate low to moderate rates of temperature increases and sea level rise, although community composition may change as a result and intertidal and shallow subtidal habitats could suffer extent losses due to coastal squeeze. These sensitivities would need to be taken into account when undertaking restoration. Also, the suitability of a given location may be influenced by potential climate change impacts, and schemes should ideally be future-proofed by considering future conditions, utilising the latest climate change projections.

Various knowledge gaps have been highlighted throughout this project. For several habitats, considerable uncertainties remain about the likely efficacy of possible marine habitat creation / restoration measures. Further trials, research and consistent monitoring are required to improve the evidence base and thus confidence in undertaking restoration. This is particularly applicable to horse mussel bed and *S. alveolata* restoration. Seagrass bed restoration could also benefit from additional, larger scale, Welsh schemes, though initial results from a 2019/20 project in West Wales appear very promising. Some mudflat and saltmarsh restoration techniques are also relatively untried in the UK, notably large scale beneficial use and sedimentation polders, and may benefit from dedicated trials. Furthermore, many ecosystem service related knowledge gaps exist, both with regard to specific native habitats (with literature relatively scarce on shellfish beds for example), and services themselves, as well as related valuations.

With regard to datalayers, several have been produced as part of this project, including two ‘opportunity’ layers which relate to the restoration of saltmarshes and mudflats, as well as native oyster habitats. Together with other layers already being produced by JNCC for horse mussels and seagrass beds, these help identify areas where some restoration activities could be undertaken and focussed.

All of these datalayers should be considered as initial aides to identifying potential locations, and the limitations which have been highlighted in Section 4 need to furthermore be taken into consideration. In addition, it is important to note that the maps do not necessarily indicate that restoration will be feasible or financially viable in a given location. The maps are intended as a focus for further discussion and investigation of the potential for restoration in some of these areas. Thus, detailed studies and local engagement will need to be undertaken before pursuing any specific locations for restoration. Relevant guidebooks should be consulted, and specialists employed where appropriate.

Whilst outside the remit of this project, some further restoration-related spatial information / datalayers could usefully be produced for the habitats which have been the focus of this review. For example, for mudflats and saltmarshes which are features of MPAs, areas of existing habitats which are in (particularly) unfavourable condition could be identified to help prioritise efforts. A similar layer is also conceivable for *S. alveolata*.

For horse mussels, where the full Welsh extent is unclear due to difficulties in mapping this often deeply subtidal habitat, further mapping effort may be required prior to identifying priority restoration areas. The opportunities layer produced by JNCC could be utilised to focus such surveying effort.

Detailed investigations into historic native oyster beds could be used to help identify historic cultch areas which could be more easily restored than bare areas. In this context the opportunities layer produced for this project could be utilised to focus surveying effort.

Further development of some of the opportunities layers could also be considered; for example, by considering further sensitivities for native oysters, notably in relation to salinity and water quality, or by excluding / highlighting urban areas in the floodplain layer developed for mudflats and saltmarshes. This could however also be achieved by overlaying existing layers when using the opportunity layers (for example Ordnance Survey products).

The layers created for this project constitute a valuable starting point for initiating more marine habitat restoration project in Wales, and ultimately facilitating the development of strategic, integrated, restoration plans which build on the four attributes of ecosystem resilience: diversity, extent, condition and connectivity.

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Appendices

Appendix A: Maps depicting the created non-opportunity datalayers

The following maps are now displayed in turn; these are all based on datalayers which have been specifically created for this project. Pleaes note that maps for the opportunity layers have not been incuded here, as these can be found in Section 4 of the main report:

- Figure A1. Welsh locations where historic saltmarsh mapping has been undertaken;
- Figure A2. Historic and current species records for *Ostrea edulis*;
- Figure A3. Historic and current species records for *Modiolus modiolus*;
- Figure A4. Historic and current species records for *Sabellaria alveolata*;
- Figure A5. Welsh maintenance dredge disposal sites, and information on depositors and material quantities and type contained within derived datalayer; and
- Figure A6. Mean (annual) wave heights and spring tide peak surface currents in Welsh waters.

Please refer to Section 2.2 of the report for detail on the datalayer creation methodology, and Section 4 for more information on the opportunity layers.

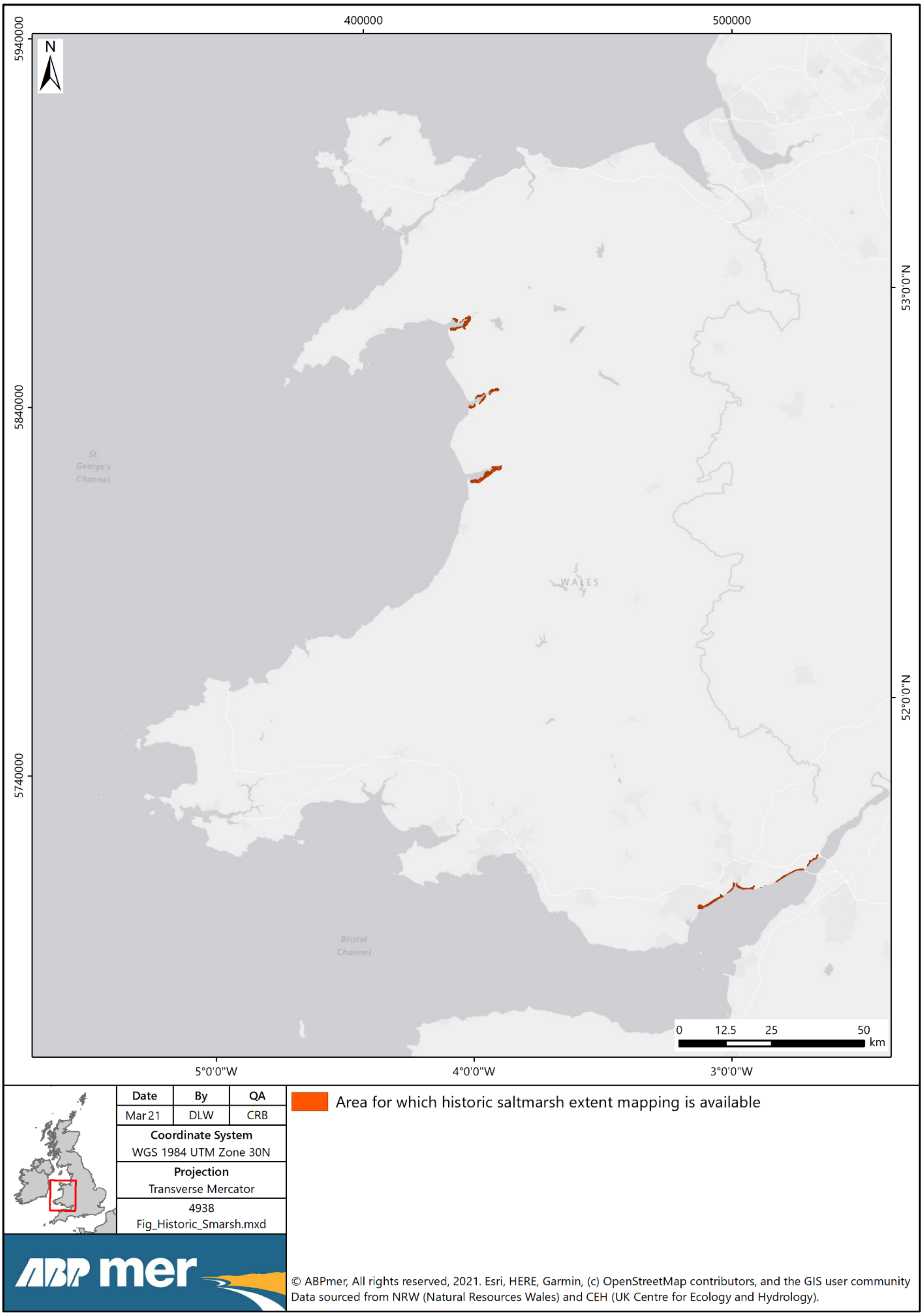


Figure A1. Welsh locations where historic saltmarsh mapping has been undertaken

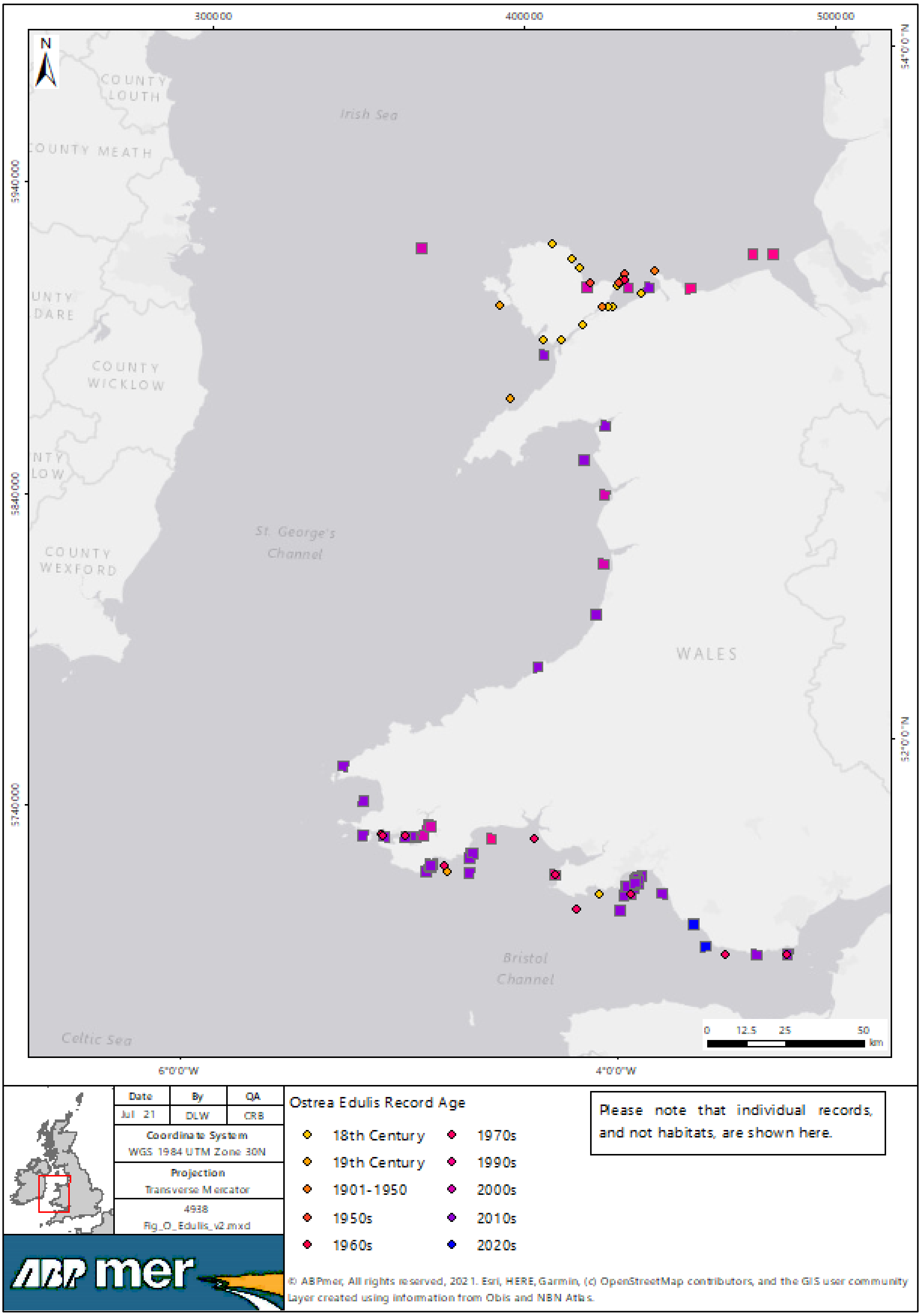


Figure A2. Historic and current species records for *Ostrea edulis* (post 1980s records shown as by 10 km^2 squares due to species sensitivity)

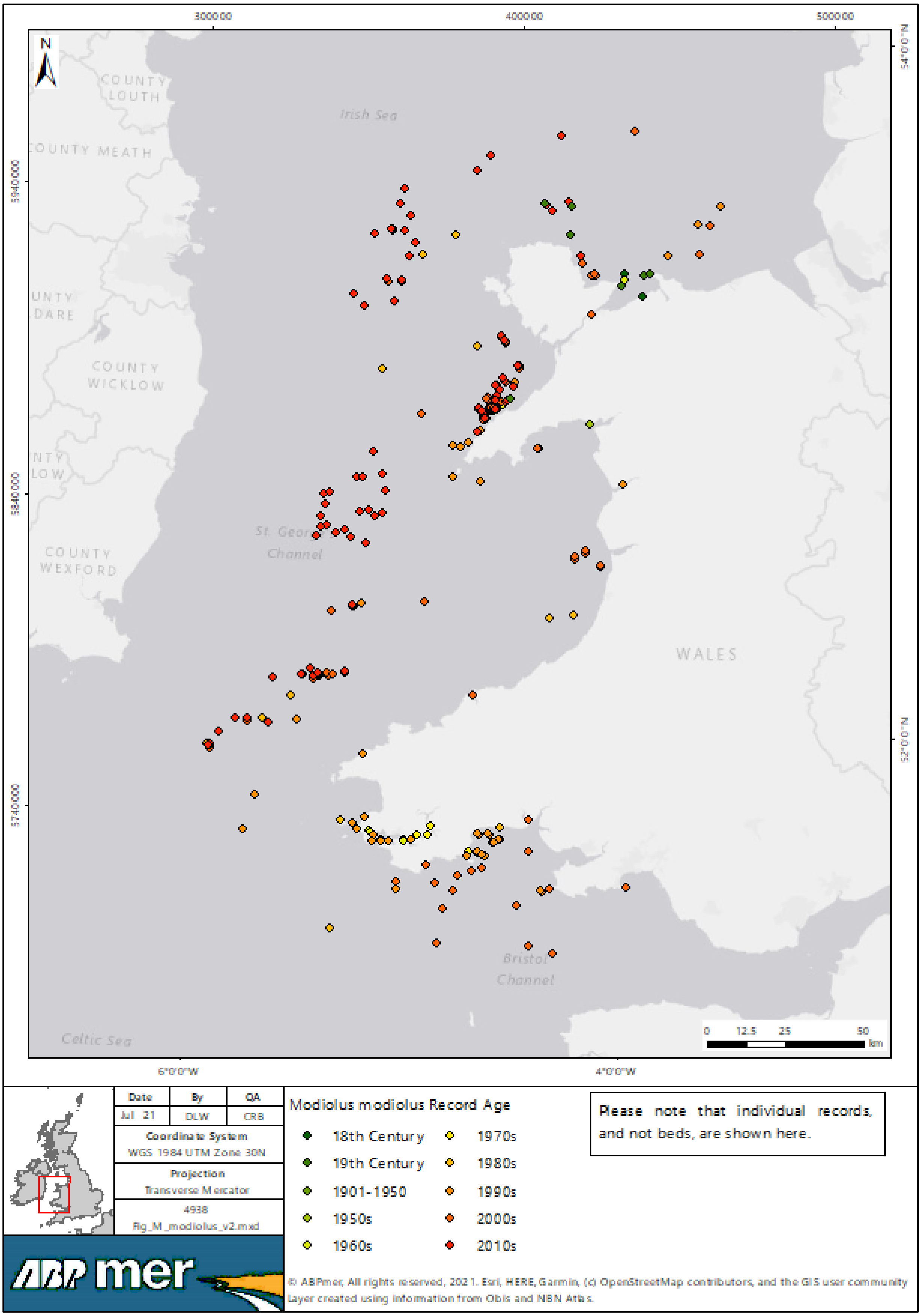


Figure A3. Historic and current species records for *Modiolus modiolus*

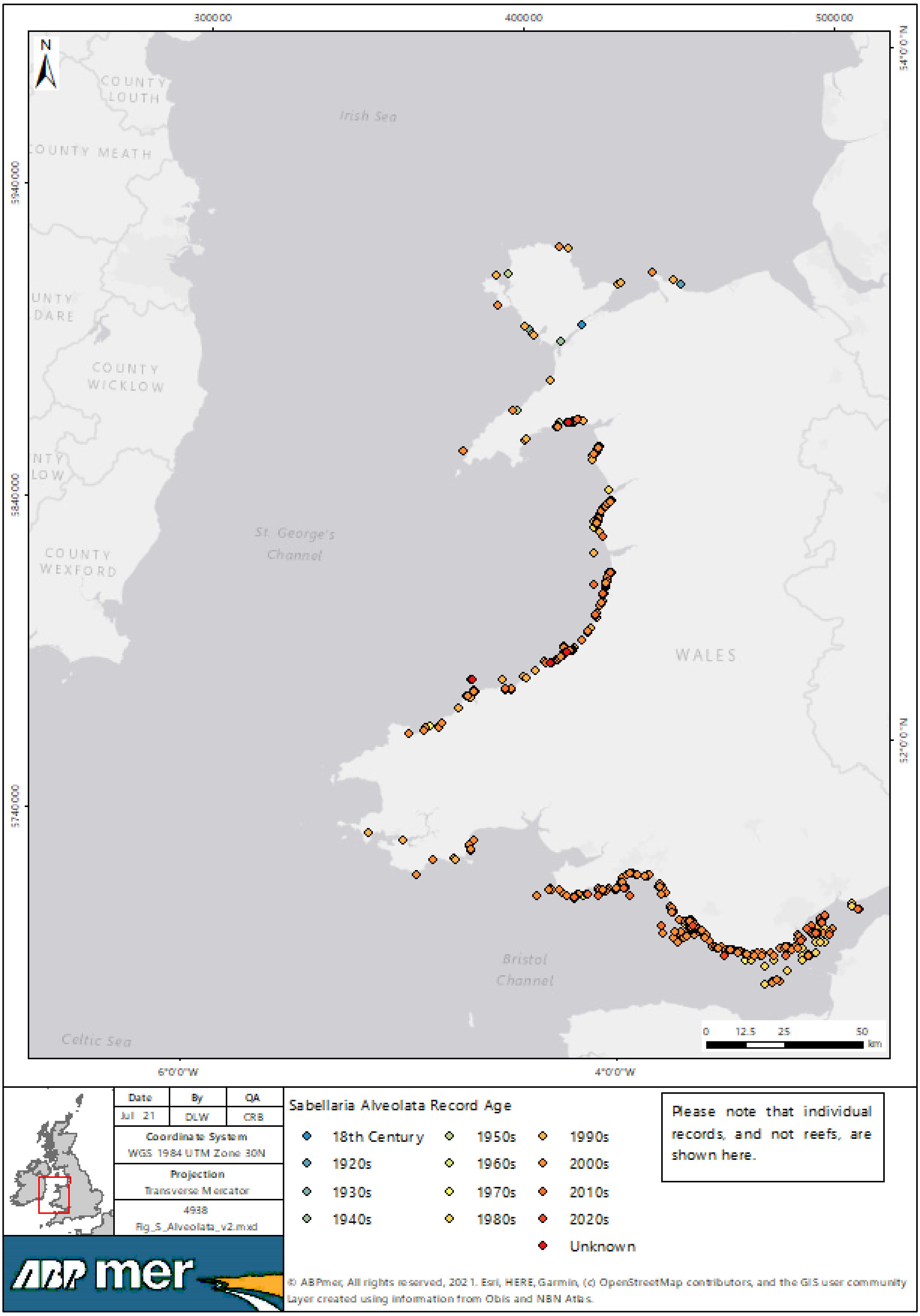
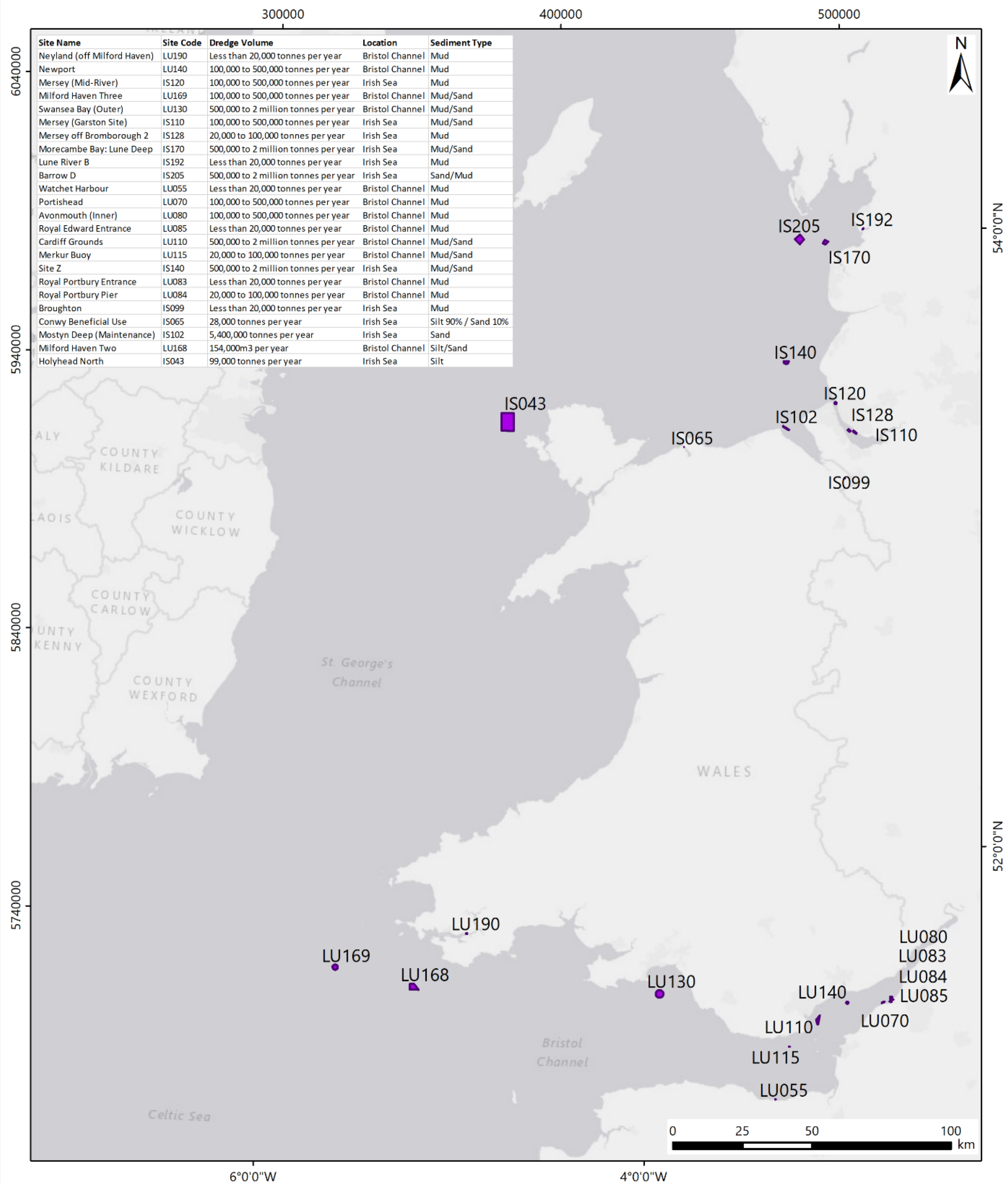


Figure A4. Historic and current species records for *Sabellaria alveolata*



Site Name	Site Code	Dredge Volume	Location	Sediment Type
Neyland (off Milford Haven)	LU190	Less than 20,000 tonnes per year	Bristol Channel	Mud
Newport	LU140	100,000 to 500,000 tonnes per year	Bristol Channel	Mud
Mersey (Mid-River)	IS120	100,000 to 500,000 tonnes per year	Irish Sea	Mud
Milford Haven Three	LU169	100,000 to 500,000 tonnes per year	Bristol Channel	Mud/Sand
Swansea Bay (Outer)	LU130	500,000 to 2 million tonnes per year	Bristol Channel	Mud/Sand
Mersey (Garston Site)	IS110	100,000 to 500,000 tonnes per year	Irish Sea	Mud/Sand
Mersey off Bromborough 2	IS128	20,000 to 100,000 tonnes per year	Irish Sea	Mud
Morecambe Bay: Lune Deep	IS170	500,000 to 2 million tonnes per year	Irish Sea	Mud/Sand
Lune River B	IS192	Less than 20,000 tonnes per year	Irish Sea	Mud
Barrow D	IS205	500,000 to 2 million tonnes per year	Irish Sea	Sand/Mud
Watchet Harbour	LU055	Less than 20,000 tonnes per year	Bristol Channel	Mud
Portishead	LU070	100,000 to 500,000 tonnes per year	Bristol Channel	Mud
Avonmouth (Inner)	LU080	100,000 to 500,000 tonnes per year	Bristol Channel	Mud
Royal Edward Entrance	LU085	Less than 20,000 tonnes per year	Bristol Channel	Mud
Cardiff Grounds	LU110	500,000 to 2 million tonnes per year	Bristol Channel	Mud/Sand
Merkur Buoy	LU115	20,000 to 100,000 tonnes per year	Bristol Channel	Mud/Sand
Site Z	IS140	500,000 to 2 million tonnes per year	Irish Sea	Mud/Sand
Royal Portbury Entrance	LU083	Less than 20,000 tonnes per year	Bristol Channel	Mud
Royal Portbury Pier	LU084	20,000 to 100,000 tonnes per year	Bristol Channel	Mud
Broughton	IS099	Less than 20,000 tonnes per year	Irish Sea	Mud
Conwy Beneficial Use	IS065	28,000 tonnes per year	Irish Sea	Silt 90% / Sand 10%
Mostyn Deep (Maintenance)	IS102	5,400,000 tonnes per year	Irish Sea	Sand
Milford Haven Two	LU168	154,000m3 per year	Bristol Channel	Silt/Sand
Holyhead North	IS043	99,000 tonnes per year	Irish Sea	Silt

Date	By	QA
Mar21	DLW	CRB
Coordinate System		
WGS 1984 UTM Zone 30N		
Projection		
Transverse Mercator		
4938		
Fig_Pot_Beneficial_Use.mxd		

Disposal sites in / near Welsh waters

Figure A5. Maintenance dredge disposal sites in/near Welsh waters, and information on depositors and material quantities and type contained within derived datalayer

Cyfoeth Naturiol Cymru
Natural Resources Wales

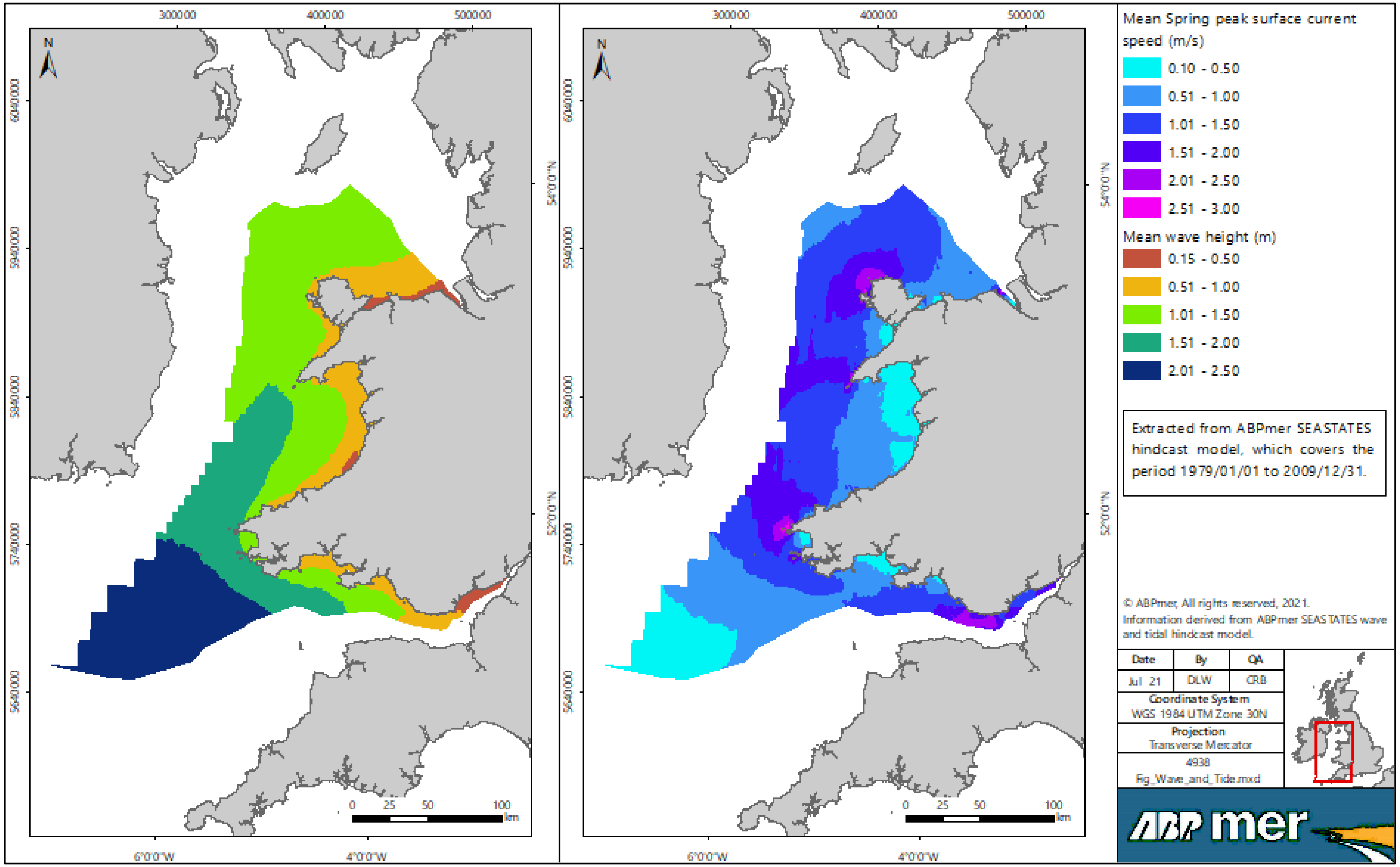


Figure A9. Mean (annual) wave heights and spring tide peak surface currents in Welsh waters

Data Archive Appendix

Data outputs associated with this project are archived in NRW's X Drive on server–based storage at Natural Resources Wales.

The data archive contains:

- The final report in Microsoft Word and Adobe PDF formats.
- A series of GIS layers on which the maps in the report are based with a series of word documents detailing the data processing and structure of the GIS layers.

Metadata for this project is publicly accessible through Natural Resources Wales' Library Catalogue https://libcat.naturalresources.wales (English Version) and https://catllyfr.cyfoethnaturiol.cymru (Welsh Version) by searching 'Dataset Titles'. The metadata is held as record no 124923.