Advice for Adaptive Environmental Management and Marine Biodiversity Enhancement Measures for Coastal Lagoon Developments

MarineSpace

Making Sense of the Marine Environment ™

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About Cyfoeth Naturiol Cymru/Natural Resources Wales

Cyfoeth Naturiol Cymru/Natural Resources Wales (henceforth referred to as NRW) is the organisation responsible for the work carried out by the three former organisations, the Countryside Council for Wales, Environment Agency Wales and Forestry Commission Wales. It is also responsible for some functions previously undertaken by Welsh Government.

NRW's purpose is to ensure that the natural resources of Wales are sustainably maintained, used and enhanced, now and in the future.

NRW works for the communities of Wales to protect people and their homes as much as possible from environmental incidents like flooding and pollution. It provides opportunities for people to learn, use and benefit from Wales' natural resources.

NRW staff work to support Wales' economy by enabling the sustainable use of natural resources to support jobs and enterprise. NRW helps businesses and developers to understand and consider environmental limits when they make important decisions.

NRW works to maintain and improve the quality of the environment for everyone, working towards making the environment and Wales' natural resources more resilient to climate change and other pressures.

MarineSpace Limited and Associates (Bright Angel Coastal Consultants Limited, Ecot Consulting Limited, and Ichthys Marine Environmental Consulting Limited) have been commissioned by NRW to conduct a review of marine biodiversity enhancement measures associated with coastal lagoon development projects.

Evidence at Cyfoeth Naturiol Cymru/Natural Resources Wales

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- Maintaining and developing the technical specialist skills of its staff;
- Securing our data and information;
- Having a well-resourced proactive programme of evidence work;
- Continuing to review and add to our evidence to ensure it is fit for the challenges facing the organisation; and
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Executive Summary

This report investigates nature conservation issues concerning the development of tidal energy lagoons. Although based on the proposed tidal lagoon in Swansea Bay, its remit is wider-ranging in order to provide strategic advice for Natural Resources Wales. It is accompanied by a separate technical report that is strictly concerned with the outputs of the Tidal Lagoon Swansea Bay project (TLSB).

There are a number of unresolved issues that relate primarily to available policy and guidance. This is especially important where development projects lead to a change in the habitat present rather than a total loss of habitat. At the moment decision-makers lack the tools to determine when changes in functionality are acceptable, or where enhancement measures that involve changing an existing habitat into a new one are consistent with the principles of sustainable management of the natural environment. Frameworks from North America offer possible models upon which to base a pragmatic approach but there is a definite need to further consider this issue and to develop appropriate guidance.

Tidal lagoons are a new concept, and although many geomorphological and ecological effects can be reliably predicted there are others where there is considerable uncertainty. In the absence of certainty an adaptive approach to management is necessary. Adaptive management plans are a relatively new concept that must be expected to evolve. This report provides an analysis of possible best practice.

A substantial part of the report involves a review of measures to create, restore or enhance critical habitats. The habitats covered are: saltmarsh and intertidal mudflats, coastal saline lagoons, intertidal sheltered muddy gravels, Seagrass beds – *Zostera* species, *Sabellaria alveolata* reefs, Native Oyster *Ostrea edulis* beds, artificial reef habitat, non-migratory fish habitat, and Atlantic Herring spawning bed habitat. As part of the review, a method has been developed for assessing levels of confidence that can be attributed to particular habitat creation measures.

It is clear that, in the majority of cases, there is an inadequate evidence base upon which to place great confidence in the measures proposed within the TLSB project. Some ideas, such as *Sabellaria alveolata* reef translocation, are novel and may prove effective after longer trials and monitoring. Others, such as the incorporation of bioblocks into the structure such as the walls of a lagoon, are dependent upon using sufficient numbers to genuinely modify the biological functionality of the structure. Considerable thought needs to be applied to the level of bioblock extent within a structure that would represent a genuine improvement in its biological performance. This is equally true of the role rock armour and gabions might play in the formation of artificial reef structures that benefit non-migratory fish, and possibly also spawning Atlantic Herring. Current proposals involving the use of granite appear to represent the weakest possible potential, based on existing knowledge of the performance of different rock types as reef structures.

Bearing in mind the expected need for extensive and repeated dredging within the lagoon basin, there is little evidence to suggest that a stable system will be established within such a structure. There will be repeated water quality and biological perturbations, even if undertaken on a rotational basis. Consequently, the potential for generating long-term functional replacement of lost habitat

within the lagoon walls is open to considerable doubt. The extent to which this may apply will depend upon local suspended sediment conditions. Consequently coastal lagoon developments built in areas where suspended sediment loads are low will require less maintenance. Conversely, those in high suspended sediment conditions, such as the Severn Estuary, are likely to require more frequent intervention. These factors mean that it would be unwise to place significant reliance upon habitat creation measures, such as laying seagrass beds within the lagoon, as realistic offsets for loss of existing habitat. Such measures in suitable locations outside the development footprint are potentially beneficial and viable under certain circumstances, subject to resolving policy issues and other considerations identified in the report.

The widely appreciated problem of invasive species in aquatic and terrestrial environments has been identified as a major factor in global biodiversity decline. Marine invasive non-native species (MINNS) are a particular problem due to the relatively open nature of the ecosystem. Vectors of invasive species include ballast water and hulls of construction and maintenance ships, contaminated shellfish species used for enhancement, and movement of invasive species, from a previously colonised location, using the lagoon wall as a ‘stepping stone’. Following wider UK and EU recognition of the MINNS problem, EU legislation and guidelines were introduced in January 2015 to promote proactive management of invasive species. These are backed by International Maritime Organisation guidelines for ballast water and biofouling management. It is recognised that biodiverse indigenous communities are better able to withstand invasive species, but that man-made structures can provide ideal habitat for MINNS. Regular disturbance (e.g. through maintenance) can lead to low diversity and favourable habitat for MINNS. Accordingly recommendations are made for biosecurity planning in-line with the new EU legislation. Measures include pre-survey; vessel audits and management; structure design (rugosity, microhabitat etc.); and post-construction/operational monitoring. Robust monitoring and close liaison with NRW, and relevant experts, are recommended to establish a proactive rather than reactive approach.

This review has highlighted the general absence of information on marine habitat creation, translocation, and restoration. There are notable exceptions such as the use of no-take zones to restore ecosystems, artificial reef construction in nearshore and offshore locations, managed realignment to yield saltmarsh, and creation of certain types of saline lagoon. Beyond this point, the knowledge gaps fall into four categories:

- Policy examples that explain how the issue of non-like-for-like mitigation and enhancement measures can be incorporated into consents for major development projects in a consistent manner;
- How ‘lifting’ anthropogenic pressures might be brought into the toolbox of measures to offset major development projects;
- The use of anthropogenic structures by non-migratory fish in nearshore and estuarine situations; and
- Factors that influence the development of mudflat in managed realignment, and possible designs to ensure mudflat rather than saltmarsh creation.

Further research in these areas would help to resolve an ongoing problem that is applicable to a wider range of development projects, including conventional and nuclear power stations, tidal lagoon projects, offshore windfarms ports and possibly other renewable energy projects.

Glossary

Abbreviation	Description	Definition
AEMP	Adaptive Environmental Management/Monitoring Programme	A structured, iterative process of robust decision making whose aim is to reducing uncertainty over time using system monitoring.
	Adverse effects	Residual negative impacts (after mitigation) upon a site or assemblage of organisms.
	Agreement of Common Ground	A clear auditable explanation of those issues where there is agreement between a developer and the Statutory Nature Conservation Advisor.
	Alternative solutions	A Habitats Directive test to determine whether there are any other feasible ways to deliver the overall objective of the plan or project which will be less damaging to the integrity of the European site affected.
	Appropriate Assessment	An evaluation of the impacts of a development project on the features of a site designated under The Habitats Directive classified under the Wild Birds Directive or listed as a Ramsar Site, in the context of the site’s conservation objectives.
	Artificial reefs	Structures created in the marine environment that lead to colonisation by marine organisms that usually occur on natural hard surfaces.
	Audit trail	A detailed record of the decision-making process.
	Biodiversity offsetting	Measures to yield biodiversity compensation to ensure that when a development damages nature (and this damage cannot be avoided or mitigated) biodiversity interest of comparable value and type will be created. They differ from other types of ecological compensation as they are normally provided distant from the development site, and can often combine compensation provision from a number of projects. Biodiversity offsetting needs to show measurable outcomes that are sustainable over time.

Birds Directive	The Wild Birds Directive or Council Directive 2009/147/EC of the European Parliament on the conservation of wild birds (the codified version of Directive 79/409/EEC as amended). The Directive provides a framework for the conservation and management of, and human interactions with, wild birds in Europe. It sets broad objectives for a wide range of activities, although the precise legal mechanisms for their achievement are at the discretion of each Member State (in the UK delivery is via several different statutes). The Directive applies to the UK and to its overseas territory of Gibraltar.
Biogenic	A substance or structure produced by living organisms or biological processes.
Borrow-dyke	A linear water-body from which material has been excavated to create a seawall (usually landward of the sea wall).
Broodstock	A group of mature individuals kept in captivity that are used for breeding purposes in aquaculture. They are often used for replacement or enhancement of shellfish seed and fish fry.
Breach	A physical break in a former sea wall intended to allow the tide to enter a managed realignment site.
Coastal Erosion	Progressive loss of foreshores and physical structures as a result of ongoing wave action.
Coastal processes	The evolution of the coast in the context of sediment transport, erosion and deposition.
Coastal squeeze	Intertidal habitat loss resulting from the high water mark being fixed by a coastal defence whilst the low water mark migrates landwards in response to sea level rise.
Coherence	The maintenance of the extent and function of the Natura 200 network, for which each individual site contributes, to ensure the necessary quality, distribution, extent and range of habitats and species.

	Compensation	Habitat creation undertaken to replace habitat lost, disturbed or damaged as a consequence of development projects, where such impacts cannot be avoided or mitigated. In the context of the EU Habitats Directive, any measure intended to offset the negative effects of a plan or project so that the overall ecological coherence of the Natura 2000 network is maintained.
CMMA	Compensation, Mitigation and Monitoring Agreement	An agreement between a developer and the statutory (and voluntary) nature conservation bodies that defines commitments to appropriate levels of compensation, mitigation and monitoring.
	Consent	Permission granted to undertake a development project.
CEMP	Construction Environmental Management Plan	Contractor generated documents will set out details of the practical execution of the construction works and the implementation of the associated environmental management measures.
	Cultch/Culch	Material that provides points of attachment for Native Oysters It usually consists of stones, broken shell and grit.
	Depensation	In population dynamics, the effect on a population (such as a fish stock) whereby, due to certain causes, a decrease in the breeding population (mature individuals) leads to reduced production and survival of eggs or offspring.
	Design objectives	The outcomes defined to ensure that a project meets its intended purpose.
	Developer	The organisation responsible for putting forward and undertaking a development project.
	Ecosystem	A community of living organisms that acts in conjunction with abiotic factors (air, water and mineral soil) to create a particular set of conditions.

EIA	Environmental Impact Assessment	The process of evaluating the likely beneficial and adverse impacts of a proposed project upon all environmental parameters, taking into account inter-related socio-economic, cultural and human-health impacts.
	EIA Directive	Environmental Impact Assessment Directive 2011/92/EU. The Directive from the European Commission that requires an EIA to be undertaken for certain projects.
EMP/EMMP	Environmental Management Plan / Environmental Management and Monitoring Plan	These are plans that detail the overall approach to implementing environmental management during and following construction of a development project.
ES	Environmental Statement	An Environmental Statement is prepared by a developer in support of certain consent applications. It summarises the findings of the as part of an Environmental Impact Assessment and is primarily intended to inform decision makers, statutory consultees, other interested organisations and members of the public about the environmental implications of the project.
ESS	Estuarine Spring Spawners	A sub-population of Atlantic Herring that spawn in estuaries between February and May.
	Evolution	The progression from one physical state to another in a sequence of almost imperceptible change.
	Extent	The absolute area of a feature. Used in this context to relate to habitat or designated site.
	Fish nurseries	Habitat that provides shelter and feeding grounds for juvenile fish.
	Flood risk management	Measures to limit the risk of flooding either by saltwater or freshwater intrusion.
FRMS	Flood risk management strategies	A set of objectives and proposed outcomes for limiting flood risk in a given section of coastline (or a river basin).
	Functionality	The contribution of particular attributes of a site to the maintenance of the features for which it is designated.

	Geogenic	A substance or structure resulting from geophysical processes.
	Habitat	The environment in which a particular kind of animal or plant usually lives. A term that is often used in the context of a particular environment dominated by certain plants that gives it a distinctive character (more correctly referred to as a biotope).
	Habitats Directive	Council Directive 92/43/EEC on the Conservation of natural habitats and of wild fauna and flora. The Directive is the means by which the European Union meets its obligations under the Bern Convention. The Directive applies to the UK and to its Overseas Territory of Gibraltar. The Habitats Directive’s function is to promote the maintenance of biodiversity by requiring Member States to take measures to maintain or restore natural habitats and wild species listed on the Annexes to the Directive at a favourable conservation status, introducing robust protection for those habitats and species of European importance. Member States are required to take account of economic, social and cultural requirements, as well as regional and local characteristics.
	Habitats Regulations	Transposition of the Habitats Directive and Wild Birds Directive into UK law.
	Intertidal	The area of land that is uncovered by the tides on a regular basis over the spring-neap cycle.
INNS/ NNS	Introduced Non-Native Species/Non-Native Species	Species that do not naturally occur within a particular geographic region that have established either as a result of deliberate measures or accidentally. Such species often arrive without the natural regulators that check their dominance and therefore out-compete existing plant and animal assemblages.
	Invertebrates	Animals without backbones, including *inter-alia* bivalve molluscs, polychaete worms and arthropods including shrimps, crabs and insects.

ITT	Invitation to Tender	The document issued by National Resources Wales when seeking tenders from consultants on its Framework contract to undertake a particular piece of work.
	Legal covenants	A promise to engage in or refrain from a specified action, secured by legally binding documentation.
	Like-for-like	Replacement of a particular area of habitat by creating new habitat that is directly analogous to that lost.
	Managed realignment	The process of creating a new sea defence line behind an existing defence line and allowing the area between the two to be inundated by creating holes (breaches) in the outer defences.
MINNS	Marine invasive non-native species	Species of plants and animals from other parts of the World whose establishment in British waters can be a problem. Such species are often introduced without natural agents of control and out-compete native species.
	Mitigation	Measures to prevent the negative impacts of a development through complete avoidance of impacts or reduction of impacts to an acceptable level. It Can include measures incorporated into the plan/project at the outset that remove any significant effect It should not be confused with compensation, which is undertaken to offset negative effects as a consequence of development projects.
	Monitoring	A process of systematic and purposeful observation to establish how project activities are progressing. It also involves the provision of feedback about the progress of the project to interested parties.
MWR	Marine Works (Environmental Impact Assessment) Regulations (as amended 2011)	The domestic legislation that transposes the EIA Directive into UK law and applies to marine licence applications for marine aggregate extraction licenses.
	National Infrastructure Planning website	This site is delivered by the Planning Inspectorate - see following.
NPPF	National Planning Policy Framework	The NPPF sets out the UK Government’s planning policies for England and how these are expected to be

		applied.
	Natura 2000	This is a network of nature protection areas in the territory of the European Union. It is made up of Special Areas of Conservation (SACs) and Special Protection Areas (SPAs) designated respectively under the Habitats Directive and Wild Birds Directive.
NSAS	North Sea Autumn Spawners	A sub-population of Atlantic Herring in the North Sea, which spawning begins around September/October in Scottish waters, and then progresses southwards through the late autumn and early winter.
NRW	Cyfoeth Naturiol Cymru/Natural Resources Wales	Cyfoeth Naturiol Cymru/Natural Resources Wales (NRW) is the organisation responsible for the work carried out by the three former organisations, the Countryside Council for Wales, Environment Agency Wales and Forestry Commission Wales. It is also responsible for some functions previously undertaken by Welsh Government. NRW's purpose is to ensure that the natural resources of Wales are sustainably maintained, used and enhanced, now and in the future.
	NERC Act 2006	Natural Environment and Rural Communities Act 2006. This domestic legislation ensures that any public body or statutory undertaker in England or Wales has consideration for conservation and biodiversity in the exercise of their functions.
	Planning Inspectorate	An executive agency of the UK Government that deals with planning appeals, national infrastructure planning applications, examinations of local plans and other planning-related and specialist casework in England and Wales.
PPS9	Planning Policy Statement 9	Sets out planning policies on protection of biodiversity and geological conservation through the planning system. These policies complement, but do not replace or override, other national planning policies and should be read in conjunction with other relevant statements of national planning policy.
	Protected species	A species of animal or plant that is protected under UK Law.
	Public Inquiry	An official enquiry into planning applications.

	Ramsar site	Wetlands of international importance, listed in accordance with the Ramsar Convention.
	Receptor sites	The site that is used to create new habitat.
	Regulators	Statutory bodies responsible for granting consents.
	Restoration	Measures to return degraded habitat to a condition that more closely approximates to a desired state.
	Seagrass	Marine flowering plants within the genus *Zostera* (in the UK) that form extensive swards in the subtidal and some intertidal situations.
	Significant effect	A term used in both Environmental Impact Assessment and in the Habitats Regulations, but with slightly different meanings. In the case of the Habitats Regulations, a 'likely significant effect' is a coarse filter to eliminate inconsequential effects but captures any measurable impacts that could possibly occur. In EIA, significance can be graded from minor to major to reflect a judgment of the magnitude of effect and the sensitivity of the receptor.
	Site integrity	The maintenance of a designated site in a state that sustains the habitats and species for which it is designated enabling their full contribution to favourable conservation status across their range, thus ensuring that condition and extent is not compromised.
SAC	Special Areas of Conservation	Sites designated in accordance with the European Council Directive 92/42/EEC (Habitats Directive) for vulnerable and threatened habitats and species listed in the Annexes of the Habitats Directive.
	Spat	Planktonic larvae at the settlement stage.
SPA	Special Protection Areas	Sites classified for their endangered and regularly occurring migratory or breeding birds in accordance with European Council Directive 2009/147/EC (Wild Birds Directive).
SSSI	Site of Special Scientific Interest	Sites designated under the Wildlife and Countryside Act (1981) as amended, on account of defined scientific interest.
SNCO/A	Statutory Nature Conservation	Bodies given a defined advisory/consultee remit under legislation. Statutory advisors provide

	Organisation/Agency	Regulators (or Competent Authorities) with reasoned assessment of the impacts of development projects so that a decision can be made using best available evidence. In Wales, the statutory advisor for nature conservation is Cyfoeth Naturiol Cymru/Natural Resources Wales (NRW).
	Sub-tidal	The sea bed below the lowest point of tidal exposure.
	Success criteria	The goals, deliverables, scope and requirements for successful completion of a project.
TLSB	Tidal Lagoon Swansea Bay	The proposed construction of a coastal lagoon development that will be used to generate energy through a series of turbines emplaced in a seawall structure connected to the coastline at Swansea Bay, Wales.
	Veliger	The planktonic larvae that is typical of some species of polychaete worms and molluscs.
WFD	Water Framework Directive	The Water Framework Directive (Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy) is a European Union directive which commits European Union member states to achieve good ecological status of all water bodies (including marine waters up to one nautical mile from shore) by 2015.

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1. Introduction and Background

1.1. Overview

Cyfoeth Naturiol Cymru/Natural Resources Wales (henceforth referred to as NRW) has been providing statutory nature conservation advice in relation to the Tidal Lagoon Swansea Bay (TLSB renewable energy project). The development within Swansea Bay, South Wales, involves construction of a 9.5 km loop of seawall connected to the coast at two ends, with in-built tidal flow power generation turbines. It is projected to encompass an area of 11.5 km^2 of intertidal and subtidal habitats.

If the project is granted consent, it will become the first commercial development of its kind in the world. There is a clear intention that this will be followed by other similar projects. An announcement on the National Infrastructure Planning website that a proposal for a location between Cardiff and Newport is expected in 2017[1] is the first in what may be a series of tidal energy lagoon projects in Wales.

Given the possibility that other projects may be proposed within Welsh waters, and might involve notified and designated nature conservation sites, NRW has commissioned studies to assist in future casework. These include:

- Definitions of mitigation and enhancement measures;
- The use of mitigation and enhancement measures outside of notified or designated sites;
- The rationale for, and best practice use of, AEMPs; and
- Viable marine habitat enhancement measures.

As part of the licensing requirements for such developments an Environmental Impact Assessment (EIA) is required. It must be compliant with The Marine Works Regulations (as amended 2011) to allow determination of any likely environmental impacts. If consent is granted for a development, predictions made during the EIA will form the basis of an Adaptive Environmental Management/Monitoring Programme (AEMP). The AEMP will be linked to construction and operational environmental monitoring as required by the relevant regulatory bodies, and detailed through project-specific licence conditions.

A draft AEMP has been submitted by TLSB (TLSB, 2014a). This requires further consideration, and development, to meet concerns raised by NRW. Considering the precedence of the TLSB project, an AEMP has never previously been used for a coastal lagoon development project. NRW therefore sees benefits in establishing a clear rationale and protocols, to enable ‘best practice’ guidance, for an effective AEMP.

The TLSB project has the potential to impose considerable changes onto the marine ecosystem of Swansea Bay. Consequently there is a need to consider measures to reduce any adverse environmental effects to an acceptable level, such that the proposed development may be granted consent. Measures to enhance biodiversity may be considered within a suite of mitigation measures, even where a development lies outside a notified or designated nature conservation site. Such measures may be especially important where project impacts involve UK Biodiversity Action Plan (UK BAP) habitats or, more recently, those listed as a habitat of principal importance in Section 42 of the Natural Environment and Rural Communities (NERC) Act, 2006. They may also be relevant in relation to the Welsh Assembly Government's obligations to report on Habitats Directive Annex I and II habitats and species, both as qualifying features of a European site, and also where present outside of the current Natura 2000 site series.

This report focuses on biodiversity issues that have been identified in the course of assessing the TLSB proposals. Its emphasis relates largely to Natura 2000 and to domestic nature protection law, and not issues relating to the Water

[1] http://infrastructure.planningportal.gov.uk/projects/wales/tidal-lagoon-cardiff/

Framework Directive (WFD). Many of the issues under consideration are, however, relevant to the WFD and aspects of this analysis may therefore be of direct, or indirect, relevance.

The viability of suggested biodiversity enhancement measures is largely untried, or little understood. There is limited evidence on which to base any judgment of likely success of deliverability of suggested biodiversity measures, either in the short or long term. NRW has therefore commissioned a critical review of the evidence base to explore the efficacy and deliverability of possible mitigation/enhancement measures within Welsh waters.

A critical aspect of development in the marine environment involves circumstances where structures provide opportunities for a wider range of organisms and communities to become established. In some cases the development structure itself may be interpreted to lead to positive outcomes. In others, 'enhancement' measures that create 'desirable' habitats may also involve the loss of existing habitats, for example the introduction of hard substratum, and associated ‘reef effects’, into a sediment-dominated ecosystem. The question therefore arises: at what point should conversion of an existing habitat to another habitat be considered either acceptable or potentially beneficial?

These issues have yet to be resolved anywhere in the UK. There is no clear guidance that might be drawn upon to assist developers or decision-makers. This current review does not extend to the philosophical issues that must be resolved in order to develop a practical approach to development in the marine environment. The analysis is inevitably constrained by the lack of a policy framework, and therefore the need to develop such a framework for informed decision-making is highlighted by its absence.

Implications for biosecurity associated with the construction and operation of coastal lagoon developments in relation to introduction and colonisation have also been reviewed.

The aim of this report is to set a clear rationale, and to inform ‘best practice’ in scoping an EIA for biodiversity enhancement measures related to coastal lagoon developments. The report also details the requirements for the development and application of an effective, fit-for-purpose, AEMP.

The report is based on a comprehensive literature review and confidence assessment of marine habitat creation and/or translocation. The assessment details the known viability and risks associated with these possible biodiversity enhancement measures. It explores aspects of their potential to offset significant adverse environmental effects of potential future coastal lagoon development projects. Within each notable habitat review, those measures that are realistically achievable are highlighted. The assessment also considers possible measures where there is no evidence of delivery, or which have previously failed.

Critically, the report builds upon ‘lessons learnt’ from the TLSB process. It is intended to provide a clear framework for interaction between NRW and developers of any future coastal lagoon development projects within Welsh waters.

2. Summary of Approach

The MarineSpace team was invited to tender for the current project, by NRW, on 10 October 2014, under the NRW Advisory Framework Lot 5: Ecological Sensitivity – Benthic habitats, benthic species and fish. The Invitation To Tender (ITT) referenced the following chapters of the Tidal Lagoon Swansea Bay Environmental Statement:

- Chapter 8 Benthic Ecology;
- Chapter 9 Fish including Recreational and Commercial Fisheries, and Appendix 7.1.1 Herring Spawning Areas;
- Chapter 23 Mitigation and Monitoring and Appendices;
- Appendix 8.3 Artificial Structures in Coastal Habitats; and
- Chapter 23.1 Adaptive Environmental Monitoring Plan.

At that time the ITT indicated that NRW required the following aspects of work:

- A critical appraisal of the biodiversity and fish enhancement measures (including biosecurity) contained within the documents referenced and examples of any existing relevant measures already in place elsewhere.
    - The likelihood of success of any of the measures proposed in terms of their, experimental nature, cost, design, and sustainability during the life of a typical tidal lagoon development (125 years); and
    - Whether any of the measures, once scaled up, would be feasible in terms of incorporating them into the design of the project and in providing biodiversity benefits.
- Proposals as to how the biodiversity enhancement measures could be strengthened. These should clearly set out recommendations for the following (to include but not be limited to the following):
    - A short list (3-5) of technically feasible and effective biodiversity and fish enhancement measures that could be used in a tidal lagoon project like TLSB;
    - Ways in which the proposed measures could be improved to increase their likelihood of success; and
    - Areas where there is a need for research to provide confidence in the efficacy of any proposed enhancement measures.
- An assessment of the definition and validity of the following terms to satisfy the requirements for biodiversity enhancement under EU, UK and Welsh legislation and policy:
    - Mitigation, compensation, enhancement, offsetting (for example, the term 'mitigation' may have different meanings under different directives and policies); and
    - The terms negative and positive effects in the ES in relation to biodiversity enhancement with reference to protected features (SAC, SPA, SSSI and S42).
- The efficacy of the use of the AEMP to detail these enhancements.

These packages of work were discussed and refined during a series of conference calls between NRW and the MarineSpace team, including a kick-off teleconference on 11 November 2014, and subsequent discussions. These established that NRW required a strategic assessment of the prospects of biodiversity enhancement, mitigation of adverse impacts, and Adaptive Environmental Management Plans in the context of possible future tidal lagoon developments within Welsh waters.

As a result of these discussions the requirements of this report were refined to encompass:

- An explanation of the context, including definitions, for biodiversity enhancement and mitigation outside of designated site boundaries. This should include a consideration of when such management options are required; and discussion of what measures are applicable and/or appropriate;
- A clear definition of Adaptive Environmental Management Plans, including rationale and best practice which needs to set a clear process for NRW to apply to future tidal lagoon development projects in Welsh waters;
- A critical review of marine biodiversity enhancement, presented for specific habitats. This needs to provide an overview of deliverability, drawing upon an extensive literature review and will include a confidence assessment;
- A discussion of how monitoring should link into the Adaptive Environmental Monitoring Plans; and
- A summary, discussing the salient findings of the review, to inform NRW staff moving forwards.

It should be noted that the habitats assessed for biodiversity enhancement measures within this report are not exhaustive. The selection of habitats covered within the report is deemed to be representative by NRW of those most likely relevant to future potential coastal lagoon projects. However it is conceivable that further marine habitats, and species, will require consideration for biodiversity enhancement, mitigation and compensation in relation to future potential coastal lagoon projects within Welsh waters.

3. Review of Definitions for Significant Effect in relation to Environmental Impact Assessment, Mitigation, Enhancement and Off-setting Outside of Statutory Designated Nature Conservation Sites

This section sets the context for the use of various terminologies likely to be associated with coastal lagoon development projects in Welsh waters. It specifically addresses points that are relevant where future projects may not be associated with statutory designated nature conservation sites, but where biodiversity enhancement measures are proposed to mitigate or off-set predicted impacts. Reference to the only existing example of a coastal lagoon development, the TLSB project (used as a case study), has been made to allow relevant and applicable comments.

3.1. Significance

It is important to bear in mind that the concept of significance differs according to individual pieces of legislation. In addition, interpretation of significance may differ according to the perceptions of organisations and individuals. Regardless of individual interpretations there are fundamental differences between the use of the term in EIA, and in the application of the Habitats Directive:

- Within the Habitats Directive, significance is used within the context of 'Likely Significant Effect', which is a very low bar: might there be an impact and would it have any potential to affect the favourable conservation status of a Natura 2000 site or the N2k network? It is a highly risk-averse process that initiates more detailed analysis and the completion of an 'appropriate assessment'. Furthermore, it is a judgment that involves the opinion of the Statutory Nature Conservation Organisation (SNCO) and the Competent Authority;
- EIA requires the determination of 'Likely Significant Effects' in which significance involves an evaluation of the possible levels of influence on particular natural attributes (based on a reasoned scientific rationale). Guidance issued by Scottish Natural Heritage (2013) provides the most comprehensive analysis, in which significance is based on:

  i. The sensitivity of the environment to change, including its capacity to accommodate the kinds of changes the project may bring about;

  ii. The amount and type of change, often referred to as the impact magnitude which includes the timing, scale, size and duration of the impact;

  iii. The likelihood of the impact occurring - which may range from certainty to a remote possibility; Comparing the impacts on the environment which would result from the project with the changes that would occur without the project- often referred to as the “do nothing" or “do minimum” comparison; and

  iv. Expressing the significance of the impacts of the project, usually in relative terms, based on the principle that the more sensitive the resource, the more likely the changes and the greater the magnitude of the changes, compared with the do nothing comparison, the greater will be the significance of the impact.

Guidance by the Welsh Government (2014) states that:

"*While the value of all the landscapes of Wales is recognised local planning authorities should have regard to the relative significance of international, national and local designations in considering the weight to be attached to nature conservation interests and should take care to avoid placing unnecessary constraints on development.*"

This approach seems to have been recognised in the matrices used by TLSB. Analysis of the EIA for TLSB must therefore be judged in this context. It is important to recognise that TLSB judgments of significance are made within the following framework:

- The affected site is not part of the Natura 2000 network (but dredge disposal may affect a N2k site);
- The affected area includes two sites with intertidal habitat designated as SSSI, outside the footprint envelope of the development (although they may be affected by the wider footprint of development activity[2]);
- The development affects several habitats considered important within the broader ambit of the Habitats Directive; and
- The development has the potential to impact upon several Section 42 (of the NERC Act, 2006) habitats and species (formerly BAP Priority habitats and species).

Importantly, however, significance in an EIA context is a judgment made by the consultant to the developer and is their professional opinion. It does not necessarily have to be agreed by the SNCO or the Competent Authority, but simply represents an evaluation that forms the basis for dialogue with the SNCO. If there is a lack of agreement on the findings of the EIA then this is a material issue that must be borne in mind by the competent authority when making its decision.

The lack of agreement between the SNCO and the developer may have considerable implications for decision-making because it is likely that conditions will be required to make sure that environmental impacts are resolved. The most effective means of documentation and resolving issues is through a 'Statement of Common Ground' and, ideally, joint signatures to a 'Compensation, Mitigation and Monitoring Agreement', which may, or may not, include the development of an Adaptive Environmental Management Plan (AEMP).

[2] In this case it is important to note that notified/designated sites outside the primary/direct impact zone of the TLSB project were considered (by NRW) to potentially be affected by the wider footprint of development activity e.g. secondary/indirect effects on habitats associated with alteration to hydrology, geomorphology, and modified coastal effects; or habitats modified due to access across a SSSI by construction equipment to access the project site. These effect pathways must be characterised and assessed rigorously for future coastal lagoon development projects.

3.2. Mitigation

Mitigation is defined by ODPM (2006) as '*measures undertaken to limit or reduce adverse effects resulting from development or other change taking place including modifications, deletions or additions to the design of the development, adaptation of methods or timing or adjustments in the nature, scale or location of the project*'.

Mitigation is an established part of the environmental assessment hierarchy that comprises: avoidance, minimization, rectification, compensation and enhancement measures. In this respect, it should be used at the minimisation and rectification stages of project design. There are, however, some obvious impacts that can only be 'mitigated' by changing design or by modifying remaining habitat with the inevitable loss of some existing ecosystem function. Some suggested examples of mitigation (not exhaustive) are provided in Table 3.2.1:

Table 3.2.1: Examples of mitigation in relation to coastal lagoon development projects (not exhaustive).

Impact	Potential modification (mitigation)	Practicality
Footprint of structure	Change the size of the structure	Only possible if there is redundancy built into the design. In the case of tidal energy lagoons there is little redundancy as this will affect power output.
	Change the position of the structure	Potentially possible providing re-location does not have other negative consequences.
Loss of biogenic reefs	Change the position of the structure	Potentially possible providing re-location does not have other negative consequences.
	Translocation	An unknown quantity with no certainty of success that also involves the loss of another biotope under the footprint of the translocation.
	Creation of a new biogenic feature (e.g. Native Oyster beds)	An unknown quantity with no certainty of success that also involves the loss of another biotope under the footprint of the translocation.

Impact	Potential modification (mitigation)	Practicality
	Design the structure to contain features that potentially replicate lost biotopes.	Potentially viable but only if carefully designed to present the same sorts of exposure, wave climate and water depth. Very strongly influenced by the choice of material and not possible to say with certainty that any particular structure will yield tangible replacement benefits.
Loss of rocky reef	Potentially possible by designing in replicate structural features of the lost reef, but this will involve an additional footprint and loss of another biotope.	Potentially viable but only if carefully designed to present the same sorts of exposure, wave climate and water depth. Very strongly influenced by the choice of material and not possible to say with certainty that any particular structure will yield tangible replacement benefits.
Loss of Atlantic Herring spawning grounds	Potentially possible by designing in replicate structural features of the lost grounds, but this will involve an additional footprint and loss of another biotope.	Potentially viable but only if carefully designed to present the same sorts of exposure, wave climate and water depth. Very strongly influenced by the choice of material and not possible to say with certainty that any particular structure will yield tangible replacement benefits.
Footprint of the tidal basin	Change the size of the structure	Only possible if there is redundancy built into the design. In the case of tidal energy lagoons there is little redundancy as this will affect power output.
Sedimentation in surrounding areas of foreshore and sub-tidally	Change the design to make sure that effects on sediment transport are minimised.	Unlikely to be completely successful because any structure placed into the water environment has inevitable consequences for both wave climes and tidal currents. Consequently some localised additional sedimentation is

Impact	Potential modification (mitigation)	Practicality
		almost inevitable.
	Structure may result in a combination of increased and reduced sedimentation in which gains and losses will 'balance out'.	Difficult to be certain and, on balance, unlikely in the case of tidal energy lagoons.
Erosion of seabed in the vicinity of the turbines and the sluices.	Probably cannot be mitigated, although erosion control using rock armour may be necessary to secure structural coherence.	Unlikely to be mitigable.
Foreshore erosion within the tidal basin.	If the foreshore is sandy then it may be possible to periodically replenish sediment, but the likelihood of this having any significant biological benefit is low.	Not likely to be mitigated in a biological context, leading to a change in biotopes present.
Sub-tidal sedimentation in the tidal basin	Periodic dredging will restore depth but will lead to the loss of the (re)established biotopes.	Cannot really be considered to be mitigation in the context of lost baseline biotopes.
Interruption of sediment supply to beaches and associated dune habitats	Possible beach feeding if a suitable source can be located.	Potentially viable but also possibly resulting in other impact footprints. No guarantee that the resulting beach will support the same biological assemblage.
Far-field effects of dredge disposal	Re-position the disposal ground.	

3.3. Offsetting

Offsetting can be defined as "…*a measurable way of ensuring that any residual damages caused by development which cannot be avoided or mitigated are made good.*" (modified from Defra, 2013).

Previous planning guidance (ODPM, 2005a, 2005b) encourages measures to offset the negative impacts of development projects, but there is no legal requirement to offset loss of undesignated biodiversity. This weakness is demonstrated by Withers (2012) who examined the effectiveness of PPS9 (ODPM, 2005a) and concluded that "… *compensation is infrequently required and, where it is, is often poorly implemented*."

At present, the only firm way of offsetting impacts on wildlife follows the provisions of the Habitats Directive and relates strictly to sites designated as Natura 2000 or to Ramsar sites. These 'offsets' are termed compensation, and there are relatively few examples of implementation in the UK (33 in England).

3.4. Enhancement

Enhancement is defined as "measures to increase the quality, quantity, net value or importance of *biodiversity or geological interest"* (ODPM, 2006). There are no absolute requirements for enhancement measures to be incorporated into development projects (SNH, 2013) although, clearly, there are examples of developments where enhancements have been incorporated into project design. SNH (2013) goes further and defines enhancement as:

"*the genuine enhancement of the environmental interest of a site or area because adverse effects are limited in scope and scale, and the project includes improved management or new habitats or features, which are better than the prospective management, or the habitats or features present there now. There is, therefore, a net or new benefit to the natural heritage*."

Examples of enhancement have a strong terrestrial bias and, as such, there is little guidance to be drawn from examples of marine works.

Unlike the terrestrial environment, where biodiversity has been reduced considerably by agriculture, the marine environment presents additional problems for developing true 'enhancement'. This is because there is almost invariably some natural biological function, even though it may be greatly modified by trawling and other extractive activities. Many of these activities are not as tightly controlled as on land, and ownership and usage arrangements are less readily modified. Thus, the most logical means of 'enhancement' is the lifting of pressures on the sea bed in the hope that biological communities will attain a higher level of productivity. Compensation for the Maasvlakte II port development at Rotterdam forms a possible analogue, in which fishing pressure has been lifted from over 24,000 ha of inshore sub-tidal habitat in order to compensate for loss of feeding grounds used by terns and Common Scoter.[3]

Enhancement that involves creation of new structures inevitably leads to a footprint within a system that provides slightly different functionality benefits. Consequently, it follows that there may be a need to make a judgment about the merits, or otherwise, of replacing one habitat with another. This is arguably a policy issue that cannot be resolved by any particular scientific argument, and is best dealt with according to prevailing pragmatism and in association with a defined research programme to determine the true benefits and drawbacks of such habitat creation.

[3] Most of the documentation for this development is available on an excellent website but is in Dutch. http://www.portofrotterdam.com/en/Port/port-in-general/Pages/maasvlakte-2.aspx

3.5. Replacing lost habitat with an alternative habitat

There are several possible scenarios:

- Structures are built on a given biotope that lead to a functional change in the associated assemblage (e.g. scour protection around the bases of wind turbine pylons);
- An existing habitat is modified to create a new desirable habitat (e.g. the sinking of the Scylla to create a new reef and diving experience); and
- Habitat translocation leads to the loss of an existing habitat that is replaced by another that is deemed to be more 'valuable'.

There appears to be no consensus on the ethics or practical outcomes of habitat modification, and the literature on the subject appears to be extremely thin, however two examples from North America provide useful thinking on the subject. Both involve fishery management but are potentially useful in developing wider thinking.

Fisheries and Oceans Canada (2002) identify a four stage hierarchy to determine the necessary measures to address habitat loss:

i. *"Create or increase the productive capacity of like-for-like habitat in the same ecological unit;*

ii. *Create or increase the productive capacity of unlike habitat in the same ecological unit;*

iii. *Create or increase the productive capacity of habitat in a different ecological unit;*

iv. *As a last resort, use artificial production techniques to maintain a stock of fish, deferred compensation or restoration of chemically contaminated sites."*

The U.S. Fish and Wildlife Service (1993) offers a slightly different approach:

i. *Category 1. "...habitat to be impacted is of high value for evaluation species and is unique and irreplaceable on a national basis or in the ecoregion section." The mitigation goal for habitat in Resource Category 1 is "no loss of existing habitat value."*

ii. *Category 2. "...habitat to be impacted is of high quality for evaluation species and is relatively scarce or becoming scarce on a national basis or in the ecoregion section." The mitigation goal for habitat in Resource Category 2 is "no net loss of in-kind habitat value."*

iii. *Category 3. "...habitat to be impacted is of high to medium value for evaluation species." The mitigation goal for habitat in Resource Category 3 is "no net loss of habitat value while minimizing loss of in-kind habitat value."*

iv. *Category 4. "...habitat to be impacted is of medium to low value for evaluation species." The mitigation goal for habitat in Resource Category 4 is "minimize loss of habitat value."*

These approaches offer ideas but do not directly translate to the situation that occurs in Wales or, indeed, elsewhere in the UK marine environment. Crucially, they highlight the need to develop a 'value' for particular habitats or species that is, arguably, assisted by BAP designations and targets. It is, however, inevitable that major developments, involving many square kilometres of sea bed will lead to substantial loss of 'extent' of existing habitat. Replacement on a 'like for like' basis is, arguably, impossible and it is unclear whether current levels of biological productivity will be maintained. They also highlight the possible benefits of developing a profit/loss account to clearly demonstrate what has happened as a result of major construction projects around the coast of Wales.

4. Rationale and Best Practice for Adaptive Environmental Monitoring/Management Programme

4.1. Introduction

This section provides an overview of the purpose, requirement and content of an Environmental Management Plan (EMP). In the case of TLSB, an Adaptive Environmental Management Plan (AEMP) has been produced[4]. The adaptive component of the plan relates to the long lifetime of the project and the potential need to modify monitoring requirements as the effects of the project unfold over time. This aspect is set out in the introductory section of the TLSB AEMP at Paragraph 1.1.05:

"*The AEMP will guide the monitoring of the effects of the Project at each stage of its progress. In the same way that results of the baseline surveys and monitoring carried out for the Environmental Impact Assessment (EIA) process have informed this document, so the results of pre-construction and construction-phase monitoring will provide up-to- date baseline data for operational-phase monitoring. During the lifespan of the Project, the AEMP will be updated, and it is for this reason that this is an adaptive plan (as noted previously). The document will continue to be updated and refined to give the best possible understanding of the Project's environmental effects enabling mitigation to be adjusted, where necessary*."

It is apparent from the above description (and the content of the AEMP) that the TLSB AEMP is essentially a monitoring plan, but with the recognition that the monitoring requirement may shift over time (i.e. some adaptation may be required).

The focus of the review of the AEMP is therefore largely restricted to considering the proposed monitoring programme in respect of structure and content as applicable to the proposed project. However, given that the AEMP, by its very title, indicates that it should be taken as the project Environmental Management Plan, it is important to consider how the AEMP would function and integrate with other components of the project's environmental management system. This aspect of the review therefore focuses on looking at the AEMP in the context of best practice relating to adaptive environmental management.

4.2. Environmental Management Planning/Plans (EMP)

4.2.1. Scope and purpose

The concept and need for project specific EMPs has been developed to deal with a criticism of the EIA process; that once approval or consent for a project is obtained the implementation of identified environmental controls is not subject to the same scrutiny. As a result, a project's construction and operational impacts may not be consistent with earlier predictions of commitments. An EMP is therefore a mechanism by which measures identified in the EIA, as necessary to avoid or minimise impacts, are detailed to facilitate implementation in the actual project.

[4] NRW's Deadline IV response on AEMP provides advice in relation to the TLSB AEMP. The information contained with the Deadline IV response relates to a Draft Interim AEMP proposal by NRW. The text provided in Section 4 of this report is intended to advise best practice and be used by NRW staff to enhance/revise the draft interim position.

EMPs are valuable tools to:

- Define details of who, what, where, and when environmental management and mitigations are to be implemented;
- Provide relevant agencies and their contractors, developers and other stakeholders better on-site environmental control over the lifetime of a project;
- Allow developers to ensure that their contractors fulfil environmental obligations; and
- Demonstrate due diligence.

There is a reliance on the EMP to ensure that a project's actual environmental impacts are consistent with those evaluated in the environmental impact assessment (EIA) process. The EMP is therefore fundamental to the EIA process, and should ensure that commitments given at a project's planning and assessment stage are carried out in the construction and/or operation stage.

An effective EMP should ensure:

- The implementation of a project's EIA, including its conditions or approval or consent;
- Compliance with environmental legislation;
- That environmental risks associated with the project are properly managed;
- Undesirable environmental effects are identified early so that management interventions can be implemented promptly to avoid major problems before they occur; and
- That project environmental objectives are fully realised.

As all projects face uncertainties with respect to design, assessment (e.g. understanding of relevant ecosystem structure and function), planning, and implementation, it is important that there is an adaptive component to the environmental management process. The primary incentive for implementing adaption within an environmental management system / plan is to increase the likelihood of achieving desired project outcomes given the identified uncertainties.

Given these uncertainties, adaptive management provides an organized, coherent, and documented process that identifies management actions in relation to measured project performance compared with desired project outcomes. Adaptive management establishes the critical feedback of information from project monitoring to inform project management, and promotes learning through reduced uncertainty.

4.2.2. Structure and Content

An EMP may take several different forms and / or comprise of a number of plans that are specific to individual topic (environmental) areas. However, it is important that there is an overarching plan that details how the overall approach to environmental management will be implemented for a project and which shows how project planning and management plans will integrate. An example is provided in **Text Box 1** for the Hinkley Point C New Nuclear power station, for which an overarching Environmental Management and Monitoring Plan (EMMP) has been produced.

Text Box 1 – Scope of the EMMP for Hinkley Point C

This **Environmental Management and Monitoring Plan (EMMP)** presents the overarching approach to environmental management and monitoring during the construction phase of the Hinkley Point C (HPC) Project. The **EMMP** makes reference to a range of supporting documentation including a series of subject specific management plans (**SSMPs**) which outline, for specific environmental topics, the environmental management measures which have been identified as mitigation requirements with the Environmental Statement for the HPC Project. The SSMPs also outline monitoring requirements which will enable the demonstration of compliance with applicable environmental quality criteria.

The **EMMP** and **SSMPs** will be used by EDF Energy and its appointed contractors who will have responsibility for work activities on the HPC development site. The **EMMP** and **SSMPs** are "live" documents that will evolve as the HPC Project progresses.

This **EMMP** (and the associated **SSMPs**) are intended to provide EDF Energy's appointed contractors with sufficient information to produce their **Construction Environmental Management Plans** (**CEMPs**) and method statements. These contractor generated documents will set out details of the practical execution of the construction works and the implementation of the associated environmental management measures.

During the project design and planning consent phase, it may be appropriate for an EMP (or EMMP) to be relatively high-level. However, it is important that an EMP at this stage sets out a credible and robust process that enables all parties to understand how environmental aspects over the life-cycle of the project will be managed. More detailed EMMPs will be required prior to construction commencing. These should be developed at detailed design stage and in conjunction with the relevant contractors and statutory authorities. If a project is approved, the conditions of approval should re-iterate the requirement for EMMPs.

Construction Environmental Management Plans (CEMPs) are typically produced prior to the construction phase and are usually contractor-led, but will require input and approval from the developer and relevant statutory bodies. A CEMP will typically comprise the following content, although this may vary from project to project, and structure will be adapted to individual requirements:

Introduction, scope and purpose;

Description of project/works – location, construction programme, equipment and plant;

Environmental Management Framework;

- Environmental policy
- Environmental aspects and impacts
- Objectives and targets
- Structure and responsibilities
- Training and awareness
- Communication
- Evaluation and monitoring
- Non-conformance and corrective actions
- Record keeping

Legal and other requirements;

Control measures (general and for individual environmental parameters);

- Incorporated into the design
- Implemented during construction

Pollution control and contingency planning.

During the operational phase of a project, the focus of an EMMP will shift towards on-going monitoring to ensure that the measures and mitigations established during construction are continuing to be effective. EMMPs covering the operational phase of a project generally draw on the measures and mitigations established in the corresponding construction EMMPs, but will also establish protocols for regular monitoring, maintenance and ensure proactive on-going management of the site.

4.3. The Role of Monitoring in Project Implementation and EMP

Environmental impact monitoring is an essential part of the EIA process, which forms part of its management component. In its most effective form, monitoring is intended to test the projected trajectory of responses to the development project and its associated environmental measures. Detailed analysis of this process is provided by Ocean Ecology (2014).

The aim of EIA is to ensure that the consequences of any development action, throughout its entire life cycle, are understood and are acceptable. Therefore EIA should have some mechanism for checks on the design, implementation, operation and decommissioning stages of the project cycle. EIA itself is a check on the project design. The implementation of monitoring and auditing is the only mechanism available to establish further checks on the later stages of the project cycle. Thus, monitoring and auditing can play a significant role in the post-decision stage of the EIA process. Money, time and effort spent on the baseline studies and predictions are all effectively rendered useless unless there is some way of testing these predictions and determining whether mitigation methods are effective and / or will have to be applied.

Monitoring in EIA usually refers to impact and mitigation monitoring. There is often significant uncertainty associated with impacts and mitigation measures, and it is therefore best-practice to undertake monitoring during both the construction and post-development phases of a project. Monitoring is essential to learn from both successes and failures:

- It is the only mechanism for comparing predicted and actual impacts and hence of checking whether mitigation measures have been put in place, testing their effectiveness and evaluating the efficiency of the project management programme;
- If mitigation measures are amenable to modification it should still be possible to reduce residual impacts identified through monitoring (i.e. there is a feedback mechanism);
- It can provide information about responses of particular receptors to impacts; and
- It can contribute to a cumulative database that can facilitate the improvement of future EIAs.

Monitoring must be closely integrated with all other project components because it is the key to the evaluation and learning components of adaptive management. An effective monitoring program is required to determine if the project outcomes are consistent with original project goals and objectives. The power of a monitoring program developed to support adaptive management lies in the establishment of feedback between continued project monitoring and corresponding project management. In order to be effective, monitoring designs must be able to discern ecosystem responses caused by project implementation (i.e. management actions) from natural variability.

Monitoring must have a clearly stated purpose against a defined hypothesis of what is expected to happen (Ocean Ecology, 2014). Setting out the purpose and defining the main reasons for undertaking measurements is the first critical step in any monitoring plan or programme. For EIA, monitoring may serve a number of purposes (Morris and Therivel, 2009):

- **Baseline monitoring** – undertaken to describe and quantify ranges of natural variation and/or directions of change that are relevant to impact prediction and mitigation;
- **Compliance monitoring** – judge whether impacts on receiving environments are compliant with numerical limits specified in permits, environmental quality standards or legislation; and
- **Impact and mitigation monitoring** – which aims to compare predicted and actual (residual) impacts, and hence to determine the effectiveness of mitigation measures.

Monitoring can also have wider benefits in providing measurement data for many other uses (e.g. enhancement of knowledge base).

Developers and relevant authorities should have a clear understanding of objectives before monitoring begins. Best practice would be to document the objectives at the start, and keep them under systematic review.

As the end result of the EIA process is to ensure that the environmental impacts of a particular scheme are considered acceptable, monitoring is required to demonstrate that the required outcomes are met (e.g. a specific level of environmental protection). The desired outcome (the proposed acceptable, residual impact) represents the influence of the presence of project infrastructure / activities, taking into account any mitigation, on an environmental receptor. It is therefore important that monitoring is sufficient to provide understanding of the individual components that contribute to the outcome (i.e. the baseline environment, the effect of the project on the baseline, and the effectiveness of mitigation).

Monitoring objectives should be developed to cover all three of the above aspects, in order to enable the overall environmental effects of the project to be understood. Detailed analysis of this approach is provided by Ocean Ecology (2014).

4.4. Summary and Recommendations

Within the context of environmental management and EIA, an EMP has a specific role to play – i.e. as a mechanism by which measures identified in the EIA as necessary to avoid or minimise impacts are detailed to facilitate implementation in the actual project.

In order to achieve this, an EMP must detail the management measures and processes that will be put in place to ensure that the desired level of environmental protection is delivered. A good EMP will therefore provide relevant project and environmental information, set out policies and procedures and implementation plans that enable the environmental components of a project to be effectively managed.

4.4.1. Clarification of monitoring need

A clear distinction needs to be made between the differing monitoring requirements of the project. As an EMP is essentially a mechanism to ensure that a project's actual environmental impacts are consistent with those evaluated in the EIA process then the focus of monitoring should be on impact verification and mitigation effectiveness. Monitoring of baseline conditions will also be required in order to provide context to the findings from the impact and mitigation monitoring.

It makes sense from a logistical perspective, and whole-project approach, to ensure that monitoring for other regulatory purposes (e.g. compliance with environmental permitting) is also built into an overall monitoring programme. This is especially relevant where monitoring overlaps or is similar to EIA / EMP monitoring requirements. However, the monitoring protocols and data required to address these needs should be clearly defined in the programme, particularly as compliance monitoring needs will be specified through appropriate permit conditions and regulatory requirements.

Similarly, monitoring to determine the effectiveness of project enhancement measures (i.e. measures that are not required to ensure that the environmental impacts of a project are acceptable) should also be detailed separately within a monitoring programme / plan. It should not, however, form part of the feedback mechanism within the EMP.

It is therefore important that the monitoring plan clearly sets out the following:

- Survey and monitoring work required to verify the predictions of the EIA process (i.e. the effects of the project). This should be linked back to a summary of the impacts defined through EIA. If baseline condition monitoring forms part of the overall verification mechanism then this should also be detailed here. In practice, verification should include an element of monitoring that enables the effects monitoring to be contextualised;
- Mitigation specific monitoring (i.e. that required to verify the effectiveness / success of discrete mitigation measures). The specific mitigation measures should be detailed along with their predicted role in respect of ameliorating environmental impact; and
- Survey and condition monitoring to establish the effectiveness of enhancement measures. While not forming a necessary part of the EMP process, it would seem logical to include all of the proposed survey and monitoring requirements in one place. Separating this aspect out will, however, make it clear that it does not need to be incorporated into the EMP process.

4.4.2. Removal of superfluous material

The report should be streamlined so that the focus is on detailing the proposed survey and monitoring requirements. The reporting of existing monitoring results, while informative, is not necessary in order to set out proposed measures for the construction and operation phases. Such material is best referenced and provided in a separate report. A description of the project would be provided in the EMP and could therefore be omitted.

4.4.3. Better definition of monitoring objectives and targets.

Monitoring objectives should link to the required need. The topic specific targets are also more akin to objectives / aims and do not provide a measurable component. A good environmental target should be a detailed performance requirement, quantified where practicable that links to the environmental objective and that needs to be set and met to achieve the objective.

4.4.4. Definition of a feedback mechanism

A description of the proposed process by which monitoring results will feed back into the overall EMP process will be needed. This is essentially the process by which adaptive management will be instigated during the project lifespan.

5. Assessment of Marine Biodiversity Enhancement Measures

It is anticipated that there will be further cases where NRW will have to advise on coastal lagoon development projects, or other major developments in the marine environment. Experience with the TLSB development project has shown that provision of replacement habitats, in-particular those not associated with any notified, or designated, nature conservation site may be necessary. This review focusses on biodiversity enhancement measures not associated with designated sites, but has also considered application to certain habitats relevant to designated sites

The TLSB development project is not associated with any European site designated under the EC Habitats Directive. The project has, however, proposed a number of biodiversity enhancement measures to ‘offset’ or 'mitigate' loss of biodiversity. The habitats that are of particular interest for the proposed TLSB development are:

- Seagrass beds associated with *Zostera* species;
- Honeycomb Worm *Sabellaria alveolata* reefs;
- Native Oyster *Ostrea edulis* beds;
- Artificial reef habitats;
- Non-migratory fish habitats; and
- Atlantic Herring *Clupea harengus* spawning beds.

This critical review considers some additional habitats that may be relevant to any future coastal lagoon development projects in Welsh waters; in-particular:

- Coastal saltmarsh;
- Intertidal mudflats;
- Intertidal sheltered muddy gravels; and
- Coastal saline lagoons.

For all of these habitats a review of the available literature has been conducted, and the feasibility of proposed biodiversity enhancement measures has been evaluated. The implications for the use of enhancement or creation of these habitats, as part of a coastal lagoon development have been assessed. It should be noted that the list of marine habitats reviewed is not exhaustive and future coastal lagoon developments may identify further habitats for enhancement, mitigation or compensation.

Some of the biodiversity enhancement methods that have been proposed (in relation to the TLSB project) include:

- Creation of artificial reef structures e.g. for intertidal biotope and fish habitat enhancement;
- Transplantation/translocation of existing habitats within the infrastructure footprint of the development e.g. moving and replanting existing seagrass beds; and
- Creation of Atlantic Herring spawning beds.

Each of these methods, where relevant, or proposed, is discussed in detail in the relevant sub-sections throughout Section 5.

5.1. Confidence assessment methodology

The feasibility and effectiveness of proposed methods of habitat creation or enhancement were subjected to a confidence assessment of the literature (evidence) base. The confidence assessment used in this document has been adapted from Kvile *et al.* (2014). It provides a semi-quantitative assessment of the quality and applicability of the current knowledge base.

The quality of the information available was scored on a scale of 1-5:

- A **score of 1** equals '**unknown**', meaning there is no literature available;
- A **score of 2** equals '**inferred**', meaning there is no literature available for the particular species or habitat in question but some level of reference can be drawn from comparable examples, such as similar species;
- A **score of 3** equals '**known**', meaning that there is some level of information available for the species or habitat in question;
- A **score of 4** equals '**well known**', meaning that there is a large amount of species- or habitat-specific information available; and
- A **score of 5** equals '**well known UK examples**', meaning that there is a high level of demonstrable information available regarding the successful enhancement / translocation of a species, or enhancement / creation of a habitat within the UK.

The quality score was applied to a series of representative factors associated with biodiversity enhancement. The factors used are:

- Directly comparable projects for habitat or species considered;
- Level of success of enhancement measures;
- Optimal environmental parameters of the habitat or species understood;
- Analogous examples;
- Temperate examples; and
- Tropical examples.

It should be noted that not all factors may be applicable to every habitat. For example, where temperate examples are available then tropical examples may not require consideration. However when temperate examples are unknown, then tropical examples have been used as a proxy where available; whilst recognising that they may not be directly applicable to effective measures for use within Welsh waters.

In the majority of cases the evidence base reviewed is not clear about the specific reasons for the enhancement being delivered. It is often unclear whether projects have been undertaken in order to offset or mitigate impacts within designated nature conservation sites, or outside of these sites. Additionally, some examples reviewed may not be directly related to enhancement or mitigation, just coincidental to the introduction of artificial substrate into the marine environment e.g. the investigation of the development of communities on seawalls or other artificial structures. Therefore, it is not possible to assign a score for a criterion related to enhancement as a result of developments/impacts that took place inside, or outside of designated sites. The evidence base does not provide the correct level of detail.

Considering that not all factors may be scored for any particular biodiversity enhancement measure, simply summing the total of all the 'factor' scores would not give a true representation of the confidence level; as different habitats may have a different number of factors considered. As a result the 'total confidence score' is divided by the number of factors scored. This results in a 'mean average confidence score'. This 'average confidence score' is used to determine varying levels of confidence for any of the biodiversity enhancement measures (habitats or species) assessed.

The confidence levels relate to ranges of the 'average confidence score' with:

- **Very Low confidence** = 'average confidence score' of **1.0-2.0**;
- **Low confidence** = 'average confidence score' of **2.1-3.5**;
- **Medium confidence** = 'average confidence score' of **3.6-4.5**; and
- **High confidence** = 'average confidence score' of **>4.5**.

A blank confidence assessment table is shown Table 5.1.1.

Table 5.1.1: Confidence assessment table.

Keys

Information quality scoring system	1 = Unknown	2 = Inferred	3 = Known	4 = Well known	5 = Well known UK examples

Confidence level	**V. Low 1-2**	**Low: 2.1-3.5**	**Med: 3.6-4.5**	**High: >4.5**

Habitat or Species Assessed		
Factors considered	**Score**	**Notes**
Directly comparable projects for habitat	1-5	
Level of success	1-5	
Environmental parameters of habitat understood	1-5	
Analogous examples	1-5	
Temperate examples	1-5	
Tropical examples	1-5	
Total score	**1-30**	
No. of factors rated	**1-6**	
Average score	**X**	

Confidence level	

5.2. Confidence assessment snapshot

The use of the assessment methodology detailed in Section 5.1 allows the level of confidence in the literature available for any proposed biodiversity enhancement measure to be scored and presented i.e. a review of the current knowledge base can be conducted and a confidence score assigned for each biodiversity enhancement measure. The individual confidence tables for each habitat reviewed in this report are presented in Section 5.3.

A summary of the overall confidence scores for each of the habitat or species enhancement measures reviewed in this report is presented in Table 5.2.1.

Table 5.2.1: The overall confidence levels for each of the habitats and species assessed in this review.

Habitat	Mean Average Confidence score	Confidence level
Saltmarsh	**5.0**	**High**
Intertidal mudflats	**4.2**	**Medium**
Coastal saline lagoons	**5.0**	**High**
Intertidal sheltered muddy gravels	**1.4**	**Very Low**
Seagrass beds	**3.6**	**Medium**
Sabellaria alveolata reefs	**2.5**	**Low**
Ostrea edulis beds	**2.4**	**Low**
Artificial substrata habitat	**3.6**	**Medium**
Non-migratory fish habitat	**2.8**	**Low**
Atlantic Herring spawning habitat	**2.0**	**Very Low**

In some cases the viability of suggested biodiversity enhancement measures is untried, or little understood, with a highly variable evidence base to judge likely success of deliverability, or long-term viability. This is especially the case considering the long-term operation (up to 125 years) of the TLSB project.

Some methods that have been considered are not deemed feasible at this point and do not have a sufficient evidence base available to be a viable option for enhancement or mitigation without further research being conducted. These methods are:

- Intertidal sheltered muddy gravels ;
- Creation of *Sabellaria alveolata* reefs; and
- Construction of Atlantic Herring spawning habitat.

5.3. Assessment of specific habitats and species which may be considered as biodiversity enhancement measures

The following sub-sections present detailed reviews and assessments of habitats and species considered for biodiversity enhancement viability in Welsh waters.

A thorough literature review has examined the habitat types required by NRW. The evidence base and literature available for each different habitat has given a confidence score to determine the efficacy as a biodiversity enhancement measure. Where no evidence is associated with a habitat, the known environmental parameters required by the habitat or species in question have been considered.

5.3.1. Saltmarsh and Intertidal mudflats

5.3.1.1. Managed realignment to create coastal saltmarsh and intertidal mudflat

Managed realignment has been used in a variety of ways to create new intertidal habitat. Early projects in England, such as Tollesbury and Orplands, simply sought to prove the concept's possibilities and practicalities. The Tollesbury site was the subject of intense monitoring and has generated a wide range of outputs, such as Garbutt and Reading (2007), and Colclough *et al*. (2006). Early projects were also evaluated by Atkinson *et al*. (2001) for their efficacy in providing feeding areas for migratory waterfowl. In subsequent years it has been used as direct compensation for port developments (Morris and Gibson, 2008) and as nature conservation enhancement. It also figures strongly in creation of more resilient flood defences (Dixon *et al.*, 2008).

In addition to deliberate realignment, there are numerous older analogues resulting from natural breaches dating back at least a century (Burd, 1994, 1995). These analogues provide a useful background to understanding how managed realignment evolves over time. In almost all cases it will become saltmarsh but there are occasional circumstances where it fails to progress to this state. These unusual circumstances occur when the relationship between sediment import and deposition is dominated by stochastic rapid export of sediment. A model for this process is described by Morris (2012).

It follows that there are certain circumstances in which managed realignment can result in the creation of mudflats, but these are unusual. More frequently, realignment will result in gradual evolution of saltmarsh (Wolters *et al*., 2008). The actual composition of saltmarsh vegetation varies hugely but it has been shown by Garbutt and Wolters (2008) that species composition takes many decades to closely mirror vegetation within 'natural' saltmarshes.

Recent analysis of managed realignment has shown that in many cases the extent of mudflat created will be very small (Mazik *et al*., 2010; Morris, 2013) and there can be no certainty that any significant extent of mudflat can be generated by conventional realignment projects. Factors that govern the evolution of realignment sites include the degree to which a site is exposed to wave action, pre-existing and designed topography and water depth within the site, and suspended sediment concentrations in the adjacent water body.

The literature on managed realignment is extensive and there are numerous case studies that can be called upon. A recent study of compensation sites for Defra, currently unpublished, includes an examination of the effectiveness of managed realignment as a compensation tool (Morris *et al.*, submitted). This study clearly highlights the difficulty of using managed realignment to create mudflat and sandflat habitat and the potential need to look at new approaches to creating this habitat. Most of the study sites involve muddy east coast sites, although there is one from a sediment deficient south coast site, and one from a sandy site on the Ribble in northwest England, that may be relevant to possible realignments in Wales.

5.3.1.1. Confidence assessment

In order to assess the confidence in the evidence presented, a confidence scoring system has been applied. The confidence table for coastal saltmarsh habitat is presented in Table 5.3.1 and for intertidal mudflat habitat in Table 5.3.2.

Table 5.3.1: Confidence assessment of viability of coastal saltmarsh habitat as a biodiversity enhancement measure.

Keys

Information quality scoring system	1 = Unknown	2 = Inferred	3 = Known	4 = Well known	5 = Well known UK examples

Confidence level	**V. Low 1-2**	**Low: 2.1-3.5**	**Med: 3.6-4.5**	**High: >4.5**

Saltmarsh		
Factors considered	**Score**	**Notes**
Directly comparable projects for habitat	5	Numerous examples of habitat creation and management in the UK, Europe and North America.
Level of success	5	Numerous examples, most of which lead to saltmarsh within 15-20 years.
Environmental parameters of habitat understood	5	Long term prognosis for almost all realignment sites is salt marsh. Determining factors are a combination of suspended sediment concentrations, topography of the realignment site, breach width and wind driven wave action.
Analogous examples	5	Numerous examples of natural breaches.
Temperate examples	5	All relevant examples are sourced from the UK or within the temperate zone.
Tropical examples	N/A	
Total score	**25**	
No. of factors rated	**5**	
Average score	**5**	

Confidence level	**High 5**

Table 5.3.2: Confidence assessment of viability of intertidal mudflat habitat as a biodiversity enhancement measure.

Keys

Information quality scoring system	1 = Unknown	2 = Inferred	3 = Known	4 = Well known	5 = Well known UK examples

Confidence level	**V. Low 1-2**	**Low: 2.1-3.5**	**Med: 3.6-4.5**	**High: >4.5**

Intertidal Mudflats		
Factors considered	**Score**	**Notes**
Directly comparable projects for habitat	5	Numerous examples of habitat creation and management in the UK, Europe and North America.
Level of success	3	Numerous examples but few successful in the long-term.
Environmental parameters of habitat understood	3	Long-term prognosis for realignment sites is usually salt marsh rather than mudflats. The factors that determine this have not been properly explored but there are theoretical models.
Analogous examples	5	Numerous examples of natural breaches.
Temperate examples	5	All relevant examples are sourced from the UK or within the temperate zone.
Tropical examples	N/A	
Total score	**21**	
No. of factors rated	**5**	
Average score	**4**	

Confidence level	**Medium 4.2**

5.3.1.2. Summary

There is a very comprehensive knowledge-base to underpin planning for creation of saltmarshes and mudflats. This evidence clearly indicates that considerable confidence can be attached to most proposals to realign sea walls to create new intertidal habitat focussing on saltmarsh. The evidence for long-term creation of mudflat is far more complicated, and in many situations the prognosis for sustainable mudflat creation is poor, although early stages of realignment evolution do yield ecologically functional mudflats.

High confidence can be attributed to the potential of managed realignment to generate saltmarsh, hence the scoring. Uncertainty about mudflat creation means that although a high score has been generated from data, the score better represents the evidence-base than the practicality of mudflat and sandflat creation, which should be investigated at a site-specific level and only after more detailed research into the mechanisms that govern the relative extent of saltmarsh and mudflat within realignment sites. Given that most realignments are less than 20 years old, any such analysis should include a major review of natural breach sites to investigate the longer-term trajectory of realignment evolution.

5.3.2. Coastal saline lagoons

The following sub-section is a direct representation of information contained within the UK BAP Saline Lagoon Working Group's 'handbook': *Saline lagoons: A guide to their management and creation* (Bamber *et al.*, 2001a).

Detailed information about saline lagoon creation and management is contained within the 'handbook' (Bamber *et al.*, 2001a), and this report will benefit little from direct reproduction of that information. Therefore when considering coastal saline lagoon habitat as a biodiversity enhancement measure associated with any proposed coastal lagoon development project, NRW staff should make direct reference to the 'handbook'.

Further detailed information about the status and water quality parameters of coastal saline lagoons is also presented by Natural England (2010). This report should also act as a reference manual for NRW staff, particularly in relation to monitoring programmes for water quality and lagoonal specialist species.

Additional evidence, subsequent to the publication of Bamber *et al.* (2001a) is detailed in Sub-section 5.3.2.8.

5.3.2.1. General Description

Coastal saline lagoons are described as areas of shallow, coastal saline (sea-) water, which are wholly or partially separated from the sea by sandbanks, shingle or, less frequently, rocks or other hard substrata. They retain a proportion of their water at low tide and may develop as brackish (hypo-saline), fully saline or hyper-saline water bodies (Bamber *et al.*, 2001a). Coastal saline lagoons often contain soft substrata that support algae and invertebrates that are not found elsewhere in Welsh waters. Additionally saline lagoons are important habitats for many bird species (JNCC, 2008). Saline lagoons are BAP habitats as well as being listed under Annex I of the EU Habitats Directive.

Ecological and geographical variations between lagoon habitats exist, and have resulted in the classification of five sub-types in the UK. Categorisation of these sub-types is on the basis of their physiography, dependent upon the nature of the separating barrier. These are:

- **Isolated lagoons**: are separated completely from the sea or estuary by a barrier of rock or sediment. Seawater enters by limited groundwater seepage or by over-topping of the sea barrier. Salinity is variable but may remain high due to water loss by evaporation;
    - **Not well suited to habitat creation**;
- **Percolation lagoons**: are separated from the sea by shingle banks. Seawater enters by percolating through the shingle or occasionally by over-topping the bank (e.g. in storms). Tidal range is normally significantly reduced, and salinity may vary. Since percolation lagoons are normally formed by natural processes of sediment transport, they are relatively transient features;

    - **Not well suited to habitat creation**;
- **Silled lagoons**: retain water at all states of the tide by a barrier of rock, termed the 'sill'. There is usually little tidal rise-and-fall and this may be out of phase with the adjacent sea. Seawater input is regular (i.e. on most tides) and although salinity may be seasonally variable, it is usually high, except where the level of the sill is near to high tide level. Distribution is restricted to the north and west of Scotland and may occur as sedimentary basins or in bedrock (termed 'oban');
    - **Not well suited to habitat creation**;
- **Sluiced lagoons**: are formed where the natural movement of water between the lagoon and the sea is modified by artificial structures, such as a culvert under a road or valved sluices. Tidal range is dependent upon the sluice efficiency and may be very low. Communities present in sluiced lagoons vary according to the type of substrate and salinity;
    - **Optimal for habitat creation**; and
- **Lagoonal inlets**: are formed when seawater enters the lagoonal inlet on each tide, i.e. an open connection to the sea (usually narrow) is present. Salinity is usually high, particularly at the seaward part of the inlet. The tidal range is usually marked and in phase with the adjacent sea. Larger examples comprise of a number of different basins, separated by sills, and demonstrate a complete gradient from full salinity through brackish to freshwater. This salinity gradient significantly increases the habitat and species diversity of the sites in which it occurs;
    - **Suited to habitat creation**.

Salinity conditions in the lagoon habitat can vary from brackish (hypo-saline owing to dilution of seawater by freshwater) to hyper-saline (i.e. greater salt concentrations than in normal seawater due to evaporation) and, as such, plant and animal communities of lagoons vary according to the physical characteristics and salinity regime of the particular lagoon (Natural England, 2010). This has resulted in significant diversity within this habitat type, with many species specifically adapted to the variable salinity regime, and a number of rare species with restricted distribution (McLeod *et al.*, 2002; Table 5.3.3).

Table 5.3.3: Species requiring specific environmental conditions which are provided within saline lagoons. (Source: Bamber *et al.*, 2001a)

Saline lagoon specialist or characteristic species	
Plants	
Lamprothamnium papulosum	Foxtail Stonewort
Tolypella nidifica	Bird's Nest Stonewort
Ruppia maritima	Beaked Tasselweed
Ruppia cirrosa	Spiral Tasselweed
Chaetomorpha linum	Wirewool Weed
Cnidaria	
Clavopsella navis	a hydroid (sea-fir)
Edwardsia ivelli	Ivell's Sea Anemone

Saline lagoon specialist or characteristic species	
Nematostella vectensis	Starlet Sea Anemone
Bryozoa	
Conopeum seurati	Lagoon Sea Mat
Victorella pavida	Trembling Sea Mat
Annelida	
Alkmaria romijni	Tentacled Lagoon Worm
Armandia cirrhosa	Lagoon Sandworm
Mollusca	
Hydrobia ventrosa	Lagoon Mud Snail
Hydrobia acuta	'Southern' Lagoon Mud Snail
Onoba aculeus	a rissoid snail
Haminoea navicula	a sea slug
Cerastoderma glaucum	Lagoon Cockle
Caecum armoricum	De Folin's Snail
Crustacea	
Cyprideis torosa	an ostracod
Idotea chelipes	Lagoon Slater
Lekanosphaera hookeri	an isopod (slater)
Gammarus insensibilis	Lagoon Sand Shrimp
Gammarus chevreuxi	a sand shrimp
Corophium insidiosum	an amphipod (*Corophium* shrimp)
Insecta	
Geranomyia bezzia	a cranefly
Glyptotendipes barbipes	a chironomid midge
Aves	
Recurvirostra avocetta	Avocet

Other species often associated with lagoons include beds of eelgrass *Zostera* spp., tasselweed *Ruppia* spp., and pondweeds *Potamogeton* spp. Rocky lagoons may also contain communities of fucoid wracks *Fucus* spp., Sugar Kelp *Laminaria saccharina*, and red and green algae. Animal communities are often characterised by mysid shrimps and other small crustaceans, worms that burrow into the sediment, gastropod molluscs, and some fish species (McLeod *et al.*, 2002).

Coastal lagoons also provide an important habitat for waterfowl, marshland birds and seabirds (UK Biodiversity Group, 1999).

5.3.2.2. UK status and distribution

Coastal lagoon habitats are a relatively uncommon habitat in the UK. The largest proportion of this habitat type occurs in England and Scotland, clustered on particular stretches of the coast. Silled lagoons are found mainly in the Outer Hebrides (termed 'Obs') and percolation lagoons occur mainly on the east coast of England, notably East Anglia, Kent and English Channel coast (Bamber and Barnes, 1996). The Solent region in southern England represents one of the highest concentrations of coastal lagoons in the UK. Four lagoon complexes within the region comprise a Special Area of Conservation (SAC) (Johnson *et al.*, 2007).

Bamber *et al.* (2001b) lists a total number of 12 saline lagoons within Wales:

- 7 are sluiced lagoons;
- 3 are lagoonal inlets;
- 1 is a percolation lagoon; and
- 1 is an isolated lagoon.

Ten confirmed saline lagoons are identified within the CCW/NRW GIS database, with a further four recorded as 'potential saline lagoon' (Table 5.3.4; Figure 5.3.1).

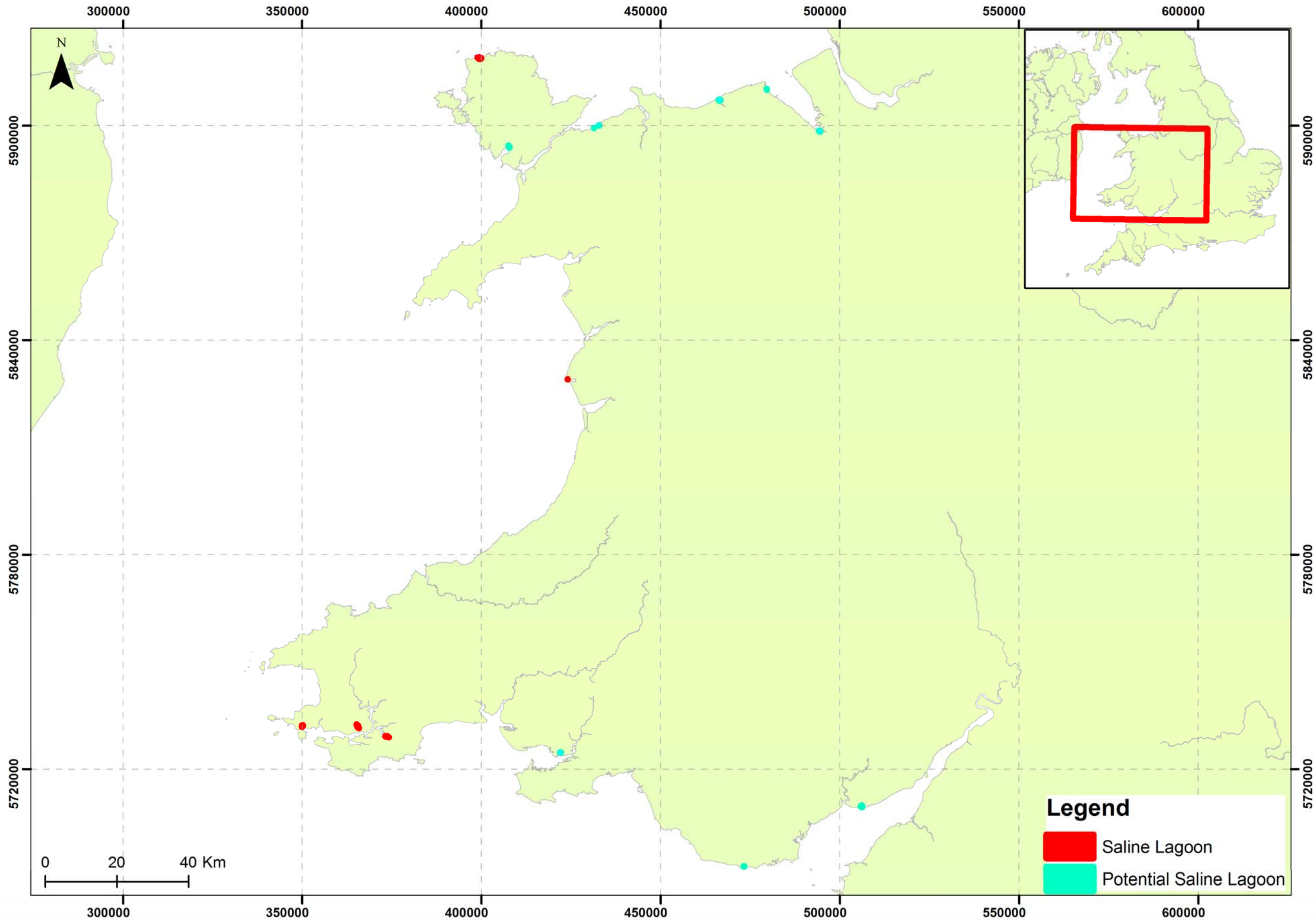
Table 5.3.4: Coastal saline lagoons in Wales. (Source: CCW/NRW GIS layer 15/01/2015 (CCW_saline_lagoons_polygon_v1))

Lagoon Name	Habitat name	Area (ha)
Cemlyn Lagoon	Saline lagoons	16.820
Rhyl Marine Lake	Potential saline lagoons	12.300
Morfa Madryn pools	Potential saline lagoons	1.364
Morfa Aber pools	Saline lagoons	0.069
Morfa Gwyllt Spit lagoon	Saline lagoons	0.365
Pickleridge lagoon, Gann flats	Saline lagoons	5.792
Neyland Weir Pool	Saline lagoons	0.974
Carew Castle Moat	Saline lagoons	7.097
Penclawydd North Pool	Potential saline lagoons	1.997
Aberthaw lagoon	Saline lagoons	1.680
Connah's Quay	Potential saline lagoons	2.149
Point of Ayr Colliery	Saline lagoons	1.214
Malltraeth Cob Pool	Saline lagoons	5.124
Gwent Levels	Saline lagoons	10.80

Three of the lagoons are designated within SACs: Cemlyn Lagoon in the Bae Cemlyn SAC, Anglesey; Morfa Gwyllt in the Pen Llyn a'r Sarnau SAC, Gwynedd; and Pickleridge Lagoon in Pembrokeshire Marine/Sir Benfro Forol SAC, Pembrokeshire.

The estimated area coverage of this habitat type has been totalled at 4,200 ha for the UK (1,200 ha for England) (UK Biodiversity Group, 1999; NBN Gateway, 2014). The extent of this habitat type present in Wales (including 'potential saline lagoons') is 67.745 ha (Figure 5.3.1).

Figure 5.3.1: Distribution of coastal saline lagoon habitat records for Wales. (Source: CCW/NRW GIS layer 15/01/2015 (CCW_saline_lagoons_polygon_v1))



5.3.2.3. Habitat requirements and creation criteria

The essence of saline lagoons is their tidal restriction, or low hydrodynamic state (Bamber *et al*, 2001a). However the exchange of saline (sea-) water into the lagoon system is considered one of the most important criteria for successful maintenance of habitat (Sheader and Sheader, 1989; Bamber *et al.*, 1992).

Optimal criteria for specialist, marine, lagoonal species are (using a hypothetical saline lagoon, according to Bamber *et al.*, 2001a):

- At least 60% of the water must persist in the lagoon at all times of the year at all states of the tide;
- Salinity variance over the range of 15-40‰;
- Sea-water input must exceed freshwater input;
- Muddy-sand substratum to sandy-mud substratum;
- Rocky substratum for specialist hard substratum biotopes;
- Shelter from wind effects;
- Up to one metre deep; and
- Have shallow margins.

The above points are most relevant to the creation of lagoon habitats, assessing condition and developing management for future lagoons (Bamber *et al.*, 2001a).

5.3.2.4. Hydrological Regime

A large number of lagoonal specialist species are closely related to fully marine rather than estuarine or freshwater species, and are tolerant of a range of salinities (Bamber *et al.*, 2001a; Natural England, 2010). Given the marine nature of these communities, freshwater inputs into the system are not considered important.

The freshwater input for coastal lagoons is usually supplied through rainwater, and surface drainage. The importance of freshwater input is site-specific; however, a freshwater supply is not necessary to a saline lagoon (Bamber *et al.*, 1992; Bamber *et al.*, 2001a; Natural England, 2010). The number of freshwater species present within the habitat can be used as an indicator of its importance.

Many coastal lagoons existing in England and Wales are transient features, with changes in salinity regimes to more freshwater environments leading to the establishment of terrestrial communities such as fen carr (UK Biodiversity Group, 1999). High freshwater inputs can result in the establishment of marsh vegetation and the loss of specialist species associated with coastal lagoons (Bamber, 1997).

Where there are hyper-salinity problems, freshwater inputs can be used to reduce salinity. In addition, in some lagoons where the seawater exchange exceeds freshwater supply, and where the lagoon outlet can readily dispense peaks of freshwater input, the salinity gradients produced are thought to assist in increasing the diversity of species found in the site (Bamber *et al*, 2001a; Natural England, 2010). As such, lagoons should be treated as site-specific, and not with general management recommendations.

5.3.2.5. Water Quality

There is significant variability (temporally and spatially) in physico-chemical conditions found within lagoons, notably pH, salinity, temperature and dissolved oxygen concentrations (Bamber *et al*, 1992; Natural England, 2010). Variability experienced within these habitats pertains to the lack of turbulent dynamics experienced as a result of the restricted tidal range.

Little work has been undertaken to investigate the impacts of changes in water quality within saline lagoons (Natural England, 2010). However, water quality issues identified from research to date include:

- Nutrient enrichment: implications for increased growth of epiphytic, floating, ephemeral benthic and phytoplanktonic algae. This has implications for lagoonal vegetation (through competition) and loss of lagoonal fauna;
- Turbidity changes: implications for light attenuation changes, smothering or inhibition of feeding lagoonal invertebrates; and
- Toxic contamination: though inputs of heavy metals, herbicides / pesticides and oil pollution into the system (Johnson and Gilliland, 2000).

Due to the limited exchange with the sea, and the reduced flushing of dissolved or suspended materials, saline lagoons are particularly sensitive to changes in nutrient loading associated with anthropogenic activities.

5.3.2.6. Soil Type

Coastal lagoons can be classified as either soft sediment or rocky lagoons (Bamber *et al.*, 2000).

Soft sediment lagoons are depositional features, with fine sediments arriving in freshwater and marine inputs. Analyses of lagoonal biotopes in England and Wales have found a single basic benthic biotope of infralittoral muddy-sand to sandy-mud (Bamber *et al.*, 2001; Natural England, 2010).

5.3.2.7. Human activities/impacts affecting habitat – management considerations

Bamber *et al.* (2001) states that when considering the sensitivity of a lagoon to impacts, or for management purposes, it is necessary to consider:

- The type of lagoon and its exchange with the sea;
- The size of the lagoon; and
- The communities and species present.

It is important to assess the ability of the lagoon to recover from an impact, how this impact is likely to affect the structural integrity of the habits, and any knock-on impacts (Bamber *et al*, 1999). Equally these factors need to be considered regarding the location and management regime required for saline lagoon habitat creation.

Threats to the integrity of the lagoon habitat in the UK, and management considerations, have been identified to include:

- Common Reed *Phragmites australis* encroachment;
- Interference with margins from grazing animals;
- Infilling from shingle bank encroachment, land reclaim or other developments;

- Degradation by excavation for shingle-bank redevelopment;
- Sea-level rise and coastal erosion;
- Abstraction or changes to freshwater inputs affecting water levels and salinity regime;
- Changes in saline water ingress into lagoon systems, altering salinity regime; and
- Input of pollution from surrounding land (nutrient enrichment).

5.3.2.8. Habitat creation, monitoring and management

Coastal saline lagoon habitat creation and monitoring is comprehensively described in chapters 4 and 5 of Bamber *et al.* (2001a) which is still the most relevant 'handbook' for this habitat. Reproduction of the 'handbook' in any synopsis is not advisable because it would miss potentially relevant information. However, as a very brief overview there are a number of criteria that should be considered when creating saline lagoon habitats:

- The lagoon should be as large as space will allow;
- If it is greater than 5 ha it should have a high shoreline-length to area ratio;
- The design should aim for a salinity of more than 20‰;
- Water depth should be 1 m or less;
- Water should enter the lagoon at a little below neap tide high water level; and
- Sluiced lagoons may provide the most reliable method of habitat creation and ease of management.

Table 5.3.5: General summary of habitat requirements.

Attributes	Requirements	Management Regime
Water quality	Saline conditions (35‰)* or conditions within 15-40‰ Sea-water exchange type specific Freshwater input (site specific)	Manage nutrient inputs which may affect the nutrient status of water inputs e.g. sewage outfalls, golf courses, horticulture, farming Maintain salinity conditions and specific exchange with the sea for each subtype
Flow regime	Maintain salinity gradient in lagoon systems identified to contain gradients	Manage water abstractions that have the potential to impact on water levels and flow regimes of freshwater into the system

*Exceptions – Tentacled Lagoon Worm and Trembling Sea-Mat (30‰)

Further relevant detailed reading can also be found Symes and Robertson (2003). Warrington *et al.* (2014) report an enhancement project undertaken at a saline lagoon in Orford Ness, Suffolk. The area had naturally developed into a coastal grazing marsh; however it was susceptible to drying out during the summer. Six shallow pools and two deeper ponds were created with the intention of attracting greater numbers of breeding birds. Two new sluices were created in order to draw water from the estuary into two new lagoons at high tide, and from there into the pools and ditches. One year after the creation of the pools the pairs of breeding wading birds had increased from 8 pairs to 23 pairs (Warrington *et al.*, 2014).

A study of a newly created coastal lagoon in Spain also examined the impact of this new habitat on the usage by waterbirds. The results showed that the lagoon contributed 26.8% of total bird abundance within the region. The habitat was considered to play a fundamental role in the enrichment of bird species in the region and is an important conservation tool (Arizaga *et al.*, 2014).

5.3.2.9. Confidence assessment

In order to assess the confidence in the evidence presented a confidence scoring system has been applied. The confidence table for coastal saline lagoon habitat is presented in Table 5.3.6.

Table 5.3.6: Confidence assessment of viability of coastal saline lagoon habitat as a biodiversity enhancement measure.

Keys

Information quality scoring system	1 = Unknown	2 = Inferred	3 = Known	4 = Well known	5 = Well known UK examples

Confidence level	**V. Low 1-2**	**Low: 2.1-3.5**	**Med: 3.6-4.5**	**High: >4.5**

Coastal Saline Lagoon Habitat		
Factors considered	**Score**	**Notes**
Directly comparable projects for habitat	5	Numerous examples of habitat creation and management.
Level of success	5	Numerous examples of successful habitat creation and on-going management.
Environmental parameters of habitat understood	5	Most of the fundamentals and detailed requirement of the multiple environmental parameters are recognised and understood. Certainly to a degree to enable delivery of successful enhancement measures.
Analogous examples	N/A	
Temperate examples	5	All relevant examples are sourced from the UK or within the temperate zone.
Tropical examples	N/A	
Total score	**20**	
No. of factors rated	**4**	
Average score	**5**	

Confidence level	**High 5**

5.3.2.10. Summary

The distribution of coastal saline lagoons and pressures affecting them are well understood for Welsh examples as well as those across the UK. This is true both for those within statutory designated nature conservation sites, and also for those occurrences outside of such sites.

The confidence in being able to create, manage and maintain coastal saline lagoons is very high. Site selection and preparation is critical, and it should be noted that site-specific investigations for suitability of creating this habitat will be required. There are many successful examples within UK estuaries, as well as at open coast sites. In the case of enhancement measures, associated with any Welsh coastal lagoon development projects, it is likely that site selection will be located within estuaries. One of the main constraints may be identifying suitable coastal habitat/land that can be used for habitat creation. This possibly limiting factor is relevant not just to coastal saline lagoons, but also intertidal mudflat and coastal saltmarsh habitat creation/enhancement schemes.

Overall, techniques for creating coastal saline lagoons are tried and tested. They may therefore be a ‘headliner’ for biodiversity enhancement measures, where like-for-like habitat creation is not essential. Further, there are targets under the UK BAP for creation of this habitat. So a win-win situation might be negotiable under some circumstances.

5.3.3. Intertidal sheltered muddy gravels

5.3.3.1. Introduction

Muddy gravels are mostly found in estuaries, drowned river valleys and sea lochs, in areas that are protected from wave action and strong tidal streams. They occur both on the shore and in the shallow infralittoral. The fauna that dominates this habitat-type consists of polychaete worms, anemones, burrowing bivalves and algae (UK Biodiversity Group, 1999).

Shores that have a freshwater influence can be expected to have lower biodiversity than fully marine habitat, although polychaeta can still dominate and bivalves such as Common Cockle *Cerastoderma edule* and Native Oyster *Ostrea edulis* can be found (JNCC, 2014c). Sheltered muddy gravels are a BAP habitat as well as being listed under Annex I of the EU Habitats Directive.

5.3.3.2. Approaches to restoration/enhancement

A thorough search of the literature has produced no results in relation to enhancement or creation of sheltered muddy gravels.

5.3.3.3. Confidence assessment

In order to assess the confidence in the evidence presented a confidence scoring system has been applied. The confidence table for sheltered muddy gravel habitat is presented in Table 5.3.7.

Table 5.3.7: Confidence assessment of viability of sheltered muddy gravel habitat as a biodiversity enhancement measure.

Keys

Information quality scoring system	1 = Unknown	2 = Inferred	3 = Known	4 = Well known	5 = Well known UK examples

Confidence level	**V. Low 1-2**	**Low: 2.1-3.5**	**Med: 3.6-4.5**	**High: >4.5**

Intertidal Sheltered Muddy Gravels		
Factors considered	**Score**	**Notes**
Directly comparable projects for habitat	1	None known.
Level of success	N/A	No examples to assess.
Environmental parameters of habitat understood	3	Basic environmental parameters are understood, although it is unlikely that they are sufficiently well known to inform viable habitat creation at this current time.
Analogous examples	1	None known.
Temperate examples	1	None known.
Tropical examples	1	None known.
Total score	**7**	
No. of factors rated	**5**	
Average score	**1.4**	

Confidence level	**Very Low 1.4**

5.3.3.4. Summary

Given the lack of any evidence of attempts to recreate sheltered muddy gravel habitat, no statement regarding the efficacy of use of this habitat as an enhancement measure can be made. It is important to note that should any coastal lagoon development project impact upon this habitat, then any proposal to recreate it has to be considered with a very low confidence. Any EIA must therefore make a full consideration of impacts, with little or no opportunity to propose residual impacts through delivery of mitigation.

If any future coastal lagoon development proposes enhancement measures associated with sheltered muddy gravel habitat, and where it is not actually impacting that habitat, then consideration could be given to proposed methods. This would be with the understanding that the enhancement measure would effectively be a pilot study, purely to determine if this habitat can feasibly be created. Considering this point, it is currently unlikely that sheltered muddy gravel habitat will be proposed as a biodiversity enhancement measure.

5.3.4. Seagrass beds – *Zostera* species

5.3.4.1. Introduction

Seagrasses, or eelgrasses, are flowering plants that form meadows or 'beds' that are found in sheltered marine and estuarine environments. Seagrass beds can exists both within the intertidal and subtidal zones, dependent upon the species present. Within the UK there are two notable species of seagrasses. These species are: *Zostera marina* Common Eelgrass which is generally a subtidal species, and *Z. noltii* Dwarf Eelgrass, a species with an intertidal distribution.

Between the 1970s-early 2000s the taxonomic status of a potential third species *Z. angustifolia* was a matter of debate, especially in relation to the plasticity of growth forms and general morphology, and also the reproductive strategies found in the wild. *Z. marina* is primarily a biennial sexually reproducing plant through flowering and dispersion of seeds, although certain populations may demonstrate some annual vegetative reproduction. In this aspect *Z. angustifolia* demonstrates a similar life strategy to *Z. marina*. In contrast *Z. noltii* displays both sexual and vegetative reproductive strategies.

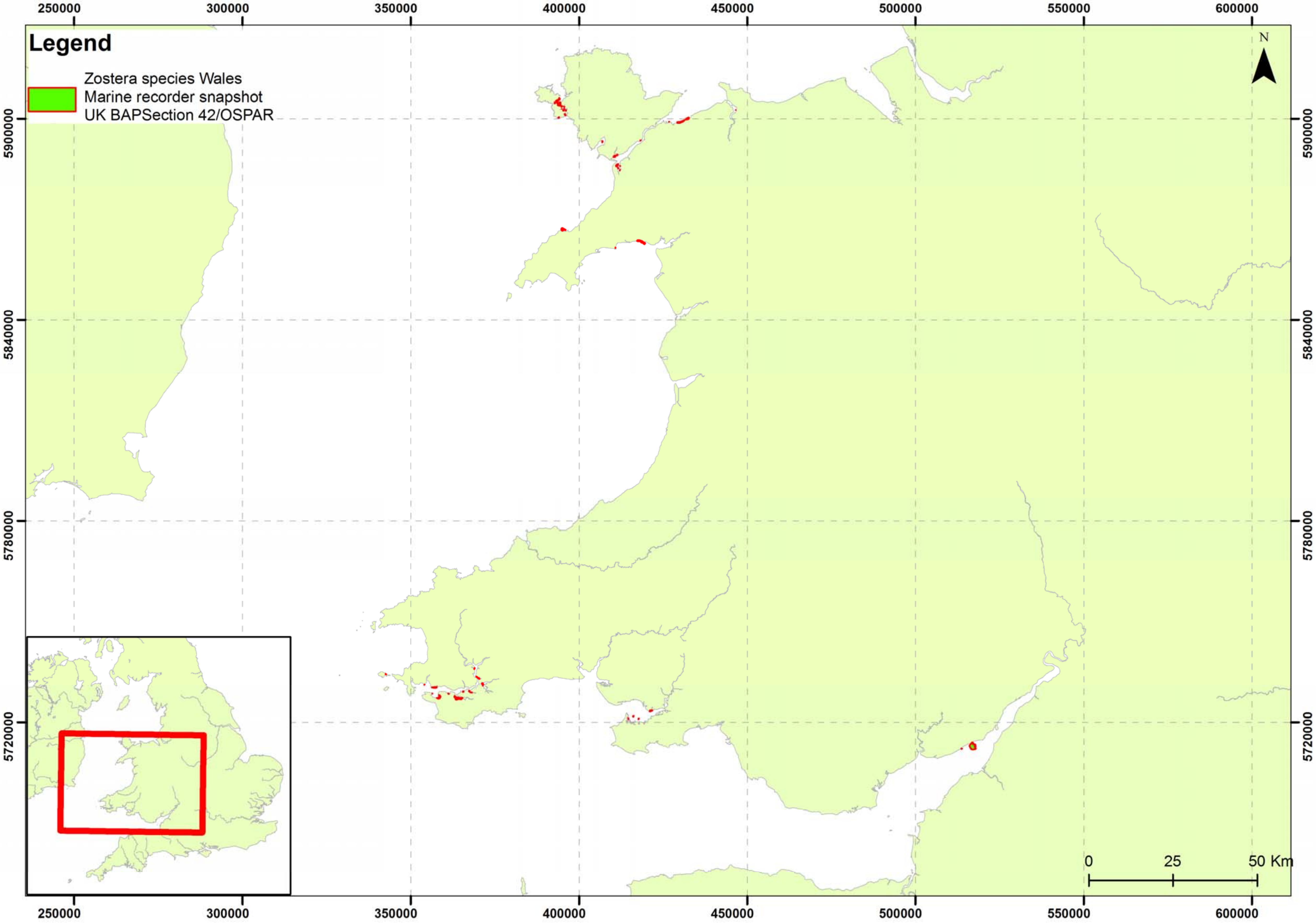
In addition to reproductive strategies, *Z. angustifolia* was recorded to be associated with intertidal habitats. It appears to demonstrate many of the same morphological characteristics as *Z. noltii*, with some notable variation associated with the apical tip of the leaf blade. Recent research by Provan *et al.* (2008) posits that *Z. angustifolia* should be considered a variant of *Z. marina*, and they refer to the primarily sexual form as *Z. marina* var. *angustifolia*.

The seagrass species that will be considered are *Z. marina* (including *Z. marina* var. *angustifolia*) and *Z. noltii* with a distribution indicated in Figure 5.3.2. It should be noted that at certain locations within the Severn Estuary a mixture of *Zostera marina* and *Z. noltii* can co-habit heterogeneous habitat consisting of sediment pockets located between boulder and cobble fields. Records of this habitat type, with a presence of either species, let alone a mosaic of both species, have an extremely restricted range in the UK (MarLIN, 2015).

Zostera spp. beds are important fish nurseries. They increase habitat structural complexity, harbour significant biodiversity and, where present, can represent a large proportion of the total marine primary production (SERG, 2014). Seagrass beds are a UK BAP priority habitat and are also listed as declining in the North Sea and Celtic Sea on the OSPAR list of threatened and / or declining species and habitats.

It is thought that a lack of genetic diversity may be contributing to the vulnerability of established beds to wasting disease. Provan *et al.* (2008) recently conducted a genotype tests to investigate the possibility of a genetic bottleneck. This analysis identified a possible bottleneck in the chloroplast genome. Extremely low levels of between-population diversity suggest that all sub-populations in the UK can be treated as a single management unit for each species.

Figure 5.3.2: Distribution of *Zostera* species records for Wales. (Source: NRW Marine Recorder snapshot 2015). Note that the image can be electronically zoomed to show polygons.



Seagrass beds are under threat from pollution and from increased sedimentation, which blocks out sunlight and hinders growth. Over the last century the UK population of *Zostera* spp. has been subject to a 'wasting disease', which resulted in decimated beds across many parts of the UK by the 1940s. It is only since the 1980s that some recovery has occurred, although there are still many notable anthropogenic pressures on eelgrass beds. These pressures include habitat loss due to infrastructure projects, physical disturbance such as trampling, dredging, anchoring, and the use of mobile bottom-trawling gear (JNCC, 2014b).

As part of the TLSB development project, encouragement of seagrass habitats within the lagoon basin is suggested. The current knowledge base for seagrass bed enhancement and restoration projects is discussed below and the feasibility of such proposals within Swansea Bay is appraised.

5.3.4.2. Approaches to restoration/enhancement

Transplanting seagrasses has been attempted worldwide, although not just for *Zostera* spp. Members of other genera, such as *Posidonia* and *Thalassia* (Thornhaug, 1985; West *et al.*, 1990; Fonesca *et al*., 1994; Jager *et al.*, 2002) have also been investigated. Transplantation methods may involve either the use of shooting plants (seedlings or mature plants), or with seeds, and has been used to restore damaged seagrass, as well as create new habitat (Lee and Park, 2008).

Bos *et al.* (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 (van Katwijk, 2003; in Bos *et al.*, 2005) with locations chosen using the following criteria:

- 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.

Further criteria were used to select transplantation areas on a local-scale (van Katwijk and Wijgergans, 2004):

- Sediment should be stable and not too coarse;
- Depth of the location should be between +15 cm above and -20 cm below mean sea-level (MSL);
- Wave exposure should be slight; and
- A thin layer of seawater should remain on the intertidal area during low tide.

Seedlings used for this transplantation project were dug up by hand at a 'source / donor' location elsewhere in the Dutch Waddenzee. Sampling of the donor seedlings was conducted every 9 m^2 to ensure genetic diversity. The donors were rinsed in seawater and transported to the transplantation site whilst maintaining the same temperature as recorded at the donor site.

The seedlings were planted in a hexagon shape with the distance between two plants remaining consistent, although low and high density planting units were investigated (Bos *et al.*, 2005). This study showed that higher densities of transplants have favourable effects on survival. Additionally, planting in open spaces within a Blue Mussel *Mytilus edulis* bed had a positive effect on transplant survival during the first growing season. This may indicate habitat security facilitated by the mussel bed which reduced exposure to wave action, and other turbation events, compared to transplanting into areas of bare sediment habitat. However, it is hypothesised that this protective mechanism may not be strong enough to support long-term survival of the eelgrass bed (Bos *et al.*, 2005).

Bos and van Katwijk (2007) describe a further study on transplanted seagrass that examined the same parameters as those described above. Their results also showed that survival was significantly higher in the high density units, compared with the low density units i.e. areas with lower distances between plantings appeared more robust than locations with wider gaps between each plant.

The effects of hydrodynamic exposure were also examined, with survival being 75%, 60% and 20% (after seven weeks) at the low, intermediate and high wave and tide exposure sites, respectively. Additionally, the study looked at the effects of seagrass planted next to a *M. edulis* bed, as in the previous study (Bos *et al.*, 2005). The investigation showed that these plants survived significantly longer, and in better condition, than transplantations located at a distance of 60 m seaward of the *M. edulis* beds.

Seagrass restoration of *Z. marina* was used to mitigate construction impact in Boston Harbour, Massachusetts (USA). Seagrass harvesting was conducted after research confirmed that the donor site was extensive and dense. Harvesting involved two techniques (Leschen *et al.*, 2009):

- Divers snapped off one shoot at a time ensuring 3-5 cm of rhizome[5] remained and shoots were then grouped into bundles of 50; and
- Divers dug small clumps of seagrass whilst leaving the sediment intact around the rhizomes and placing the clumps in dive bags.

Planting for this project was conducted by a combination of hand and frame-planting, and seed dispersal. Frame-planting involved the use of weighted wire mesh frames to which paired seagrass shoots were tied, with 50 shoots per frame. 1m^2 frames with string grids were also used to overcome some problems associated with the wire frames. For seed dispersal flowering shoots were collected and maintained in sea water tanks until seeds dropped, approximately 300,000 seeds were planted either by divers scratching seeds into sediment or broadcasting seeds from a boat (Leschen *et al.*, 2009).

Transplantation was followed by monitoring of survival rates and surveys of the biological composition of the established beds. After two years the expansion observed was significant and coverage was over 2 hectares. The transplanted seagrass bed approached or exceeded local natural seagrass beds in population structure, abundance and diversity (Leschen *et al.*, 2009).

Many of the methods that have historically been used to transplant seagrass are considered to be costly and labour intensive. For example, divers are often required (Fonseca *et al.*, 1998). A study that compared the use of a mechanised planting boat or manual transplanting found that using the mechanised system did not significantly improve the planting process compared with manual planting. Differences in survival were not statistically significant, although the mean survival

[5] The rhizome is a horizontal stem extension below the sediment surface that extends the seagrass and connects neighbouring plants.

associated with manual planting was higher. Poorer initial planting success meant that the mechanised system required greater labour input (Fishman *et al.*, 2004).

Lee and Park (2008) describe an alternative method that does not require divers to plant the transplants. In this case, oyster shells were used as anchoring devices for the transplants. Planting consisted of two *Z. marina* shoots attached to an oyster shell dropped from a boat. The transplants using this method successfully established, and the survival rates were comparable with those seen in other common planting techniques. The results suggest that the physiological status of the transplants was similar to that of the natural population after seven months. A similar method was used to transplant *Z. marina* in China, where several rhizome shoots were bound to a single elongate stone using biodegradable thread (Zhou *et al.*, 2014). However, these transplants were buried by hand in the intertidal zone. The method led to high success, with a survival rate of >95% after three months. The resulting seagrass beds were monitored and, after becoming established, were considered equal to natural beds in terms of shoot height, biomass and seasonal variations.

Pickerell *et al.* (2005) developed a method for dispersing *Z. marina* seeds that has the potential to facilitate large-scale restoration. Their method involved mesh nets suspended by buoys that were stocked with mature reproductive shoots of seagrass collected during the second week of seed release. As the seeds ripen they are naturally released from the net and they fall to the bottom and germinate. A survey of the seedlings indicated that the recruitment rate using this method was approximately 6.9%. This method is considered to be low-cost, effective and simple and consequently has a significant impact on establishment of plants from seeds.

A seagrass restoration project in Chesapeake Bay, USA was conducted for *Z. marina*. Early projects at this site involved transplantation of *Z. marina* from healthy populations to restoration areas. This, however, proved to be cost and labour intensive as well as being potentially detrimental to donor beds. Seeds were therefore collected by SCUBA diving or mechanical processes and two seed dispersal methods were trialled. The dispersal methods (spring seed buoys and fall seed broadcasts) were evaluated for cost effectiveness (Busch *et al.*, 2010). The spring seed buoy method involved transferring seeds into mesh bags and was based upon the method described by Pickerell *et al.* (2005). The fall seed broadcast method involved the use of a mechanical seed sprayer to disperse the seagrass seeds. All seed broadcasts took place in October before the ambient water temperatures dropped below 15°C, prior to *Z. marina* seed germination (Busch *et al.*, 2010). The results of the analysis showed that the cost per seed for the spring buoy method and the fall broadcast method was $0.10 and $0.22 respectively. It is considered that these methods of seed dispersal are competitive in terms of cost compared to more traditional restoration methods used (Busch *et al.*, 2010).

Techniques for restoring *Z. marina* seagrass beds were examined by Marion and Orth (2010), including mechanical harvesting and seed dispersal techniques. They also evaluated storage conditions for seagrass seeds. Their results showed that mechanical harvesting was an effective approach for collecting seeds and resulted in low impacts on donor beds. A comparison of seed dispersal techniques found that deploying seed bearing shoots in buoys produced fewer seedlings and required more effort compared with isolating, storing and broadcasting seeds in the fall. It was also found that viable seeds can be separated and stored and survival during storage can be as high as 95% over 3 months. The study therefore concluded that there is potential for expanding the scale of seed based *Z. marina* restoration (Marion and Orth, 2010).

5.3.4.3. Confidence assessment

In order to assess the confidence in the evidence presented a confidence scoring system has been applied. The confidence table for seagrass beds is presented in Table 5.3.8.

Table 5.3.8: Confidence assessment of viability of *Zostera* species beds as a biodiversity enhancement measure.

Keys

Information quality scoring system	1 = Unknown	2 = Inferred	3 = Known	4 = Well known	5 = Well known UK examples

Confidence level	**V. Low 1-2**	**Low: 2.1-3.5**	**Med: 3.6-4.5**	**High: >4.5**

***Zostera* spp. seagrass beds**		
Factors considered	**Score**	**Notes**
Directly comparable projects for habitat	4	Well known for *Zostera marina*.
Level of success	4	Well documented success using a variety of methods, but predominantly associated with labour intensive hand planting either in the intertidal, or in the subtidal using divers.
Environmental parameters of habitat understood	4	The environmental parameters required for the establishment and maintenance of transplanted/artificially created *Zostera* spp. beds are well understood and parameterised for a wide selection of case studies.
Analogous examples	3	*Z. marina* is considered analogous for *Z. noltii*. See factor below also. However, considering the intertidal nature of *Z. noltii* the analogous nature between species has been scored slightly conservatively.
Temperate examples	3	Considering the biogeographic range of *Zostera* spp. the evidence reviewed demonstrates a bias to examples from temperate ecosystems which are relevant analogues for Welsh waters. Consideration of the heterogeneous nature of the sediment pockets within a cobble and boulder nature have resulted in a slightly conservative scoring of this factor as most examples are for homogenous sediment habitats.
Tropical examples	N/A	
Total score	**20**	
No. of factors rated	**5**	

Average score	**3.6**	

Confidence level	**Medium 3.6**

5.3.4.4. Summary

The evidence base is moderately strong regarding the creation of *Zostera* spp. seagrass beds as a biodiversity enhancement measure. The confidence assessment assigns a medium confidence, with the lack of significant UK examples preventing a confidence score of high.

There is a large body of information available for restoration and creation of *Zostera* spp. seagrass habitats. Many of the projects described in this sub-section have proven successful in introducing seagrass, or restoring depleted habitats, both in the UK and worldwide, and the environmental parameters of functional beds are well understood. This provides a good evidence base which can be used to inform decisions for any future coastal lagoon developments that may be proposed within Welsh waters. It should be noted, however, that at certain locations within the Severn Estuary a mixture of *Zostera marina* and *Z. noltii* can co-habit heterogeneous habitat consisting of sediment pockets located between boulder and cobble fields. In these circumstances any proposal or ES must give full consideration to this mosaic habitat.

As most Welsh *Zostera* beds are already designated or qualifying features of marine protected areas careful consideration to the sourcing of seeds or rhizomes will be required. Harvesting within existing European sites and SSSIs will likely be prohibited, and certainly permissions will be required if removal is deemed viable. Any proposal, ES, and certainly an AEMP, must make explicit consideration of this point.

Correct identification of a suitable location within proximity to any proposed coastal lagoon development is fundamental to success. Consideration of water quality parameters and the high likelihood of maintenance dredging during the lifespan of these types of developments mean that establishing *Zostera* spp. within the lagoon basin, as a biodiversity enhancement measure, may be highly questionable.

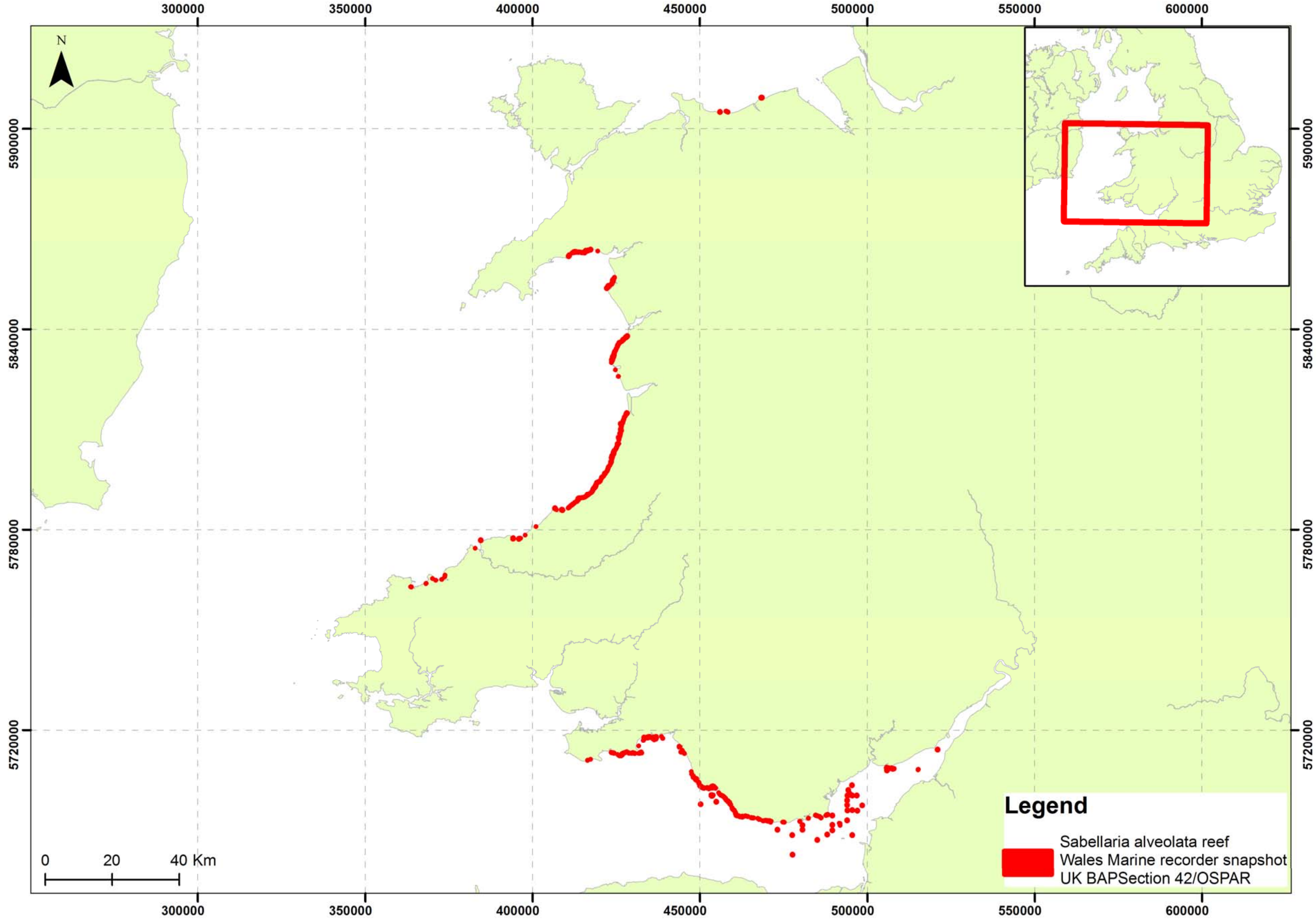
An effective ES and AEMP will need to make full consideration and assessment of these likely constraints, in order to deliver a robust assessment allowing high confidence that *Zostera* spp. beds can be viable as a biodiversity enhancement measure; certainly within a development's lagoon basin footprint.

5.3.5. *Sabellaria alveolata* reefs

5.3.5.1. Introduction

The Honeycomb Worm *Sabellaria alveolata* is a tube-dwelling species of polychaete worm that is found in the intertidal, and occasionally subtidally, in areas with high nearbed flow rates and high sandy sediment bedloads (Jackson, 2008). Notable examples of subtidal reef are associated with the Severn Estuary, and offshore from Spurn Head at the mouth of the Humber Estuary. In its more common intertidal habitat, *S. alveolata* is found attached to hard substrata on exposed coasts with moderate to high water movement, where sand is available for tube-building (Jackson, 2008).

Figure 5.3.3: Distribution of *Sabellaria alveolata* reef habitat records in Wales. (Source: NRW Marine Recorder snapshot 2015) Note: records are indicative of reef habitat, not just presence of the worm itself.



S. alveolata, as with most sabellarid polychaetes, dwells within tubes which it builds from sand grains. Unlike the UK's other species of sabellarid, *S. spinulosa*, which commonly aggregates as a few individual worms or disaggregated low-elevation veneers, *S. alveolata* commonly builds extensive reefs within the intertidal zone. There is a high organisational level to the reef structure with the sand tubes closely aggregated such that they resemble a honeycomb, hence the species common name (Holt *et al.*, 1998). Each tube shares communal walls with its neighbours, and each has a short projection or porch, over the entrance to the tube (Egerton, 2014). This species is recorded as constructing large reefs that can be several metres across, and up to a metre proud of the attachment substrate (Holt *et al.*, 1998).

Where the reefs are relatively stable the structural complexity of the biogenic habitat can facilitate functional habitat space for epiphytes, infauna, and epifauna, aswell as far mobile species such as crustaceans.

S. alveolata is a BAP species and is also listed as an Annex I reef habitat under the Habitats Directive. The TLSB project has proposed the translocation of *S. alveolata* reefs associated with the potential development footprint. The west coast of the UK, including the Welsh coastline, is a mainstay of this species' distribution, therefore *S. alveolata* reef habitat is considering likely to interact with future potential coastal lagoon development projects in Welsh waters (Figure 5.3.3).

A literature search has not produced any substantive results for creation or translocation of *S. alveolata* reefs. However the TLSB project has investigated translocation of this habitat type at the proposed location of the potential development. Trials were at a small, demonstration scale, compared to the full-scale required to mitigate impacts should the development proceed.

5.3.5.2. Environmental parameters

S. alveolata is usually found in intertidal areas on open coasts with moderately strong to strong tidal currents and can be found at depths of up to 10 m (Jackson, 2008). The worm is associated with hard substrata such as bedrock, cobbles, pebbles and small to large boulders. Following initial settlement veliger larvae usually settle on pre-existing formations (UK Biodiversity Group, 1999). In addition to the hard substrate required for settlement, *S. alveolata* also requires a supply of suspended sand-sized sediment and therefore local hydrological climate must have sufficient energy to hold sand grains in suspension (UK Biodiversity Group, 1999). As well as association with estuarine conditions, this species is also found in areas of full salinity on open coasts (Jackson, 2008).

The reproductive behaviour of *S. alveolata* makes the hydrodynamics an important regulating factor in dispersal and recruitment. *S. alveolata* release larvae which spend between 6 weeks and 6 months in the plankton, eventually settling on existing colonies/reefs, or their dead remains (UK Biodiversity Group, 1999). This may indicate a positive chemotaxis response of veliger larvae, possibly responding to biomarkers associated with proteins secreted by the worms when bonding sand particles when tube-building.

A study of *S. alveolata* in the Bay of Mont-Saint-Michel, France, indicates that the hydrodynamic regime allows for larval retention within the bay and favours larval exchange between reefs (Ayata *et al.*, 2009). Additionally, this study noted the importance of the tidal conditions with the optimal time for spawning being during a neap tide and the importance of meteorological conditions, with certain wind directions being favourable for dispersal (Ayata *et al.*, 2009).

Sea and air temperature are believed to affect the growth of *S. alveolata*, and to affect mortality of individuals. The metabolism of the species, and the associated growth, of *S. alveolata* increases with temperature, with a plateau evident at a temperature of 20°C. Below 5°C the growth of this species becomes constrained and the worms often do not survive long periods of low temperatures, especially if they are located high up the shore (Egerton, 2014). Mass mortality events have been recorded during cold winters, especially where ice on the shore has been recorded (Holt *et al.*, 1998). As such the UK west coastline is believed to form the northern-most biogeographical range for this species.

S. alveolata is a filter feeder and requires suspended food in the surrounding water. It is therefore associated with exposed coastal conditions and turbulent high current velocity waters created by wave or tidal actions. Such conditions provide enough water movement to supply suspended food particles, and when located in close proximity to a supply of sand also provide suitable tube-building material. Both the food supply and sand particles are needed for the colonisation and growth of *S. alveolata* and development of reefs (Egerton, 2014).

5.3.5.1. Approaches to restoration/enhancement

An extensive literature search has been conducted and no evidence was found for translocation of *S. alveolata*, or other Sabellarid reefs, apart from the case study presented in the TLSB project's AEMP Revision 3. TLSB has investigated translocation of existing *S. alveolata* reef within the Swansea Bay system which is directly associated with the proposed development's footprint. This case study is discussed in detail later in this sub-section.

The literature search identified a small amount of evidence about translocation of another species of polychaete worm, *Serpula vermicularis*. *S. vermicularis* is a fan worm which lives within a calcareous tube worm and can be found attached to hard substrata such as rocks, stones, bivalve shells, and the hulls of ships, at depths of 0-250 m. This species is found in sheltered waters where the tubes can aggregate together to form small reefs (Hill, 2006). *S. vermicularis* requires completely different environmental parameters to those optimal for *S. alveolata*, however consideration of a *S. vermicularis* translocation project may provide useful insight.

S. vermicularis was collected and replanted in two Scottish sea lochs (Hughes *et al.*, 2008). Transplanted tubes were found to have gradually disappeared over the monitoring period and only remnants were left ten months after the initial translocation. This study suggests that losses may have been due to predators within the area, rather than as a result of physiological barriers. However, this experimental translocation highlights some of the difficulties that a restoration programme may face. These results suggest that future translocation projects should consider securely anchor transplanted worms and possibly enclose them within cages to decrease the rate of loss and increase the likelihood of success of re-establishment (Hughes *et al.*, 2008).

The western landfall of the proposed TLSB development's seawall will directly overlap an intertidal area at Port beach, located at the east of the mouth of the River Tawe. This shore supports a heterogeneous mosaic of communities including notable *S. alveolata* reef habitat (ES Chapter 8, Sections 8.5.8 and 8.5.9, TLSB, 2014b). Translocation of *S. alveolata* reef is proposed to reduce the impact of removal and substratum loss (ES Section 8.7, TLSB, 2014b). Acknowledging that translocation of *S. alveolata* reef has not been attempted before, the residual effects of the EIA are predicted as major adverse in an appropriately precautionary manner.

The project's AEMP Revision 3 details an investigation into translocation of existing reef at Port beach (the donor site) to a location further to the west of the River Tawe, Swansea Bay (AEMP Sections 7.3.5.3-7.3.5.36). This receptor site supports existing *S. alveolata* reef. The pilot study translocation was undertaken on 06 June 2014, so some initial statements can be made about success or failure of the exercise. However, this also means that there is effectively no time series upon which to robustly judge the success or failure of the experiment. It should be born in mind that the *Serpula vermicularis* case study showed that translocated specimens/reefs were moribund or dead within a period of ten months following translocation. A much longer time series of monitoring is therefore required to give any confidence in the efficacy of mitigation measures.

The TLSB translocation pilot study is relatively simple in its premise, facilitated by the nature of the *S. alveolata* reef habitat present at the development site. The reef habitat in Swansea Bay is associated with boulders and cobbles colonised by the worm tubes, forming sheets around the core geogenic structure. This effectively means that 'blocks' of reef (30 cm square-sized boulders) can be lifted from the donor site (the impact zone) and physically moved to the receptor site, where they can be positioned at an optimal height on the shore of the receptor site.

Reef condition at both sites was assessed to allow monitoring of reef blocks once translocated. A representative set of control reef blocks were also monitored. Whilst these reef blocks were not translocated, they were handled in that same way and intensity as those blocks moved between sites. These controls represent the pressure of handling the blocks and any compression or abrasion effects that may result from 'man-handling'.

The results of the initial pilot study showed that in general, the *S. alveolata* reef at the receptor site appeared healthier reef following a period of five weeks after translocation. Reef scoring follows the principles described in Egerton (2014) (Table 5.3.9).

Table 5.3.9: Reef descriptions used in the Tidal Lagoon Swansea Bay translocation pilot study. (From: Egerton, 1984)

Formation	Description
Patchy	Small crusts or mounds which are less than 30 cm^2
Hummock	Raised mound which are greater than 30 cm^2
Sheet	Flat crust which are greater than 30 cm^2
Reef	Large mounds which are greater than 1 m^2

A higher frequency of 'crisp apertures' (Egerton, 2014) was noted than prior to translocation, whilst a greater percentage of 'sheet' reef was found at the donor site. Both sites contained 'dead' areas of *S. alveolata*, but a higher proportion was recorded at the donor site and a lower proportion of newly settled *S. alveolata* was seen compared to the receptor site.

The results also indicated that translocation of *S. alveolata* reef blocks to the receptor site was successful, with all translocated specimens surviving. However, it is important to consider that this was only following a five week period. It is also important to note that densities in control samples, i.e. those handled but not translocated showed lower densities of tube. This suggests that the handling of the reef blocks and the physical process of relocation (disturbance) has a potentially negative effect on the reef handled.

The difference in recovery between the control samples and those translocated does not appear to have been analysed in a statistical manner. The variations are likely to be attributable to the conditions present at the donor site compared with the receptor site. Baseline characterisation appears to indicate that in general the donor site's reefs were 'healthier' than those at the donor site. It may be that environmental parameters are more favourable at the receptor site, and that the translocated reef blocks are responding to more favourable or optimal habitat conditions. However, without any statistical rigour, this is hypothetical, but an important observation nonetheless.

Considerations of potential changes in environmental conditions at the receptor site, as a result of secondary environmental effects from the presence of the lagoon structure, are not presented. It is possible that alteration of hydrological and geomorphological conditions, including changes to nearbed sediment transport, may affect survival/viability of a translocated reef at the receptor location. For any future lagoon development projects these secondary effect pathways will have to be considered when screening the suitability of viable receptor locations. Section 7.3.5.20 of the AEMP Revision 3 mentions that another monitoring survey was to be conducted in October 2014; however these results are not available for analysis/assessment at this time.

5.3.5.2. Confidence assessment

In order to assess the confidence in the evidence presented a confidence scoring system has been applied. The confidence table for *S. alveolata* reef is presented in Table 5.3.10.

Table 5.3.10: Confidence assessment of viability of *S. alveolata* reef as a biodiversity enhancement measure.

Keys

Information quality scoring system	1 = Unknown	2 = Inferred	3 = Known	4 = Well known	5 = Well known UK examples

Confidence level	**V. Low 1-2**	**Low: 2.1-3.5**	**Med: 3.6-4.5**	**High: >4.5**

Honeycomb Worm *Sabellaria alveolata* Reef		
Factors considered	**Score**	**Notes**
Directly comparable projects for habitat	3	There is only one example of translocation of *S. alveolata* reef, and that represents a small-scale and short-term pilot study located in Swansea Bay, Wales.
Level of success	2	The only known example only monitored the study habitat five weeks following translocation. This period is not long enough to allow any confident statement to be made regarding success of the measure.
Environmental	3	The autecology of *S. alveolata* reef habitat is

parameters of habitat understood		reasonably well understood.
Analogous examples	3	There is only one example in the literature base where another reef-building polychaete worm has been translocated, and that was unsuccessful.
Temperate examples	3	The two examples reviewed are both temperate and one of those is from Swansea Bay, Wales.
Tropical examples	1	No tropical examples were found.
Total score	**15**	
No. of factors rated	**6**	
Average score	**2.5**	The medium confidence score may not be conservative enough, considering that the single example of translocation of *S. alveolata* reef habitat, known in the world, has only one post-translocation monitoring event reported, and that was only five weeks following the event. However, the translocated reef blocks were in healthy condition at the receptor site.

Confidence level	**Low 2.5**

5.3.5.3. Summary

The evidence base is weak for effective translocation of *S. alveolata* reef habitat as a biodiversity enhancement measure. Actual creation of the habitat has no evidence base at all and should effectively be considered not possible. Within the time span of potential new coastal lagoon development projects within the next several years in the UK, there should be no confidence in *S. alveolata* reef habitat creation itself. Therefore the focus of habitat enhancement measures for *S. alveolata* reef should focus on translocation of existing reef structures from within an impact zone and relocating to a new viable receptor site.

The confidence assessment assigns a low confidence score for translocation. There is only a single example of translocation of *S. alveolata* reef habitat known in the world, and that has only one post-translocation monitoring event reported. Further, the monitoring results reflect the condition of translocated reef following a period of only five weeks following the relocation event. It should be recognised that one would reasonably expect a time-series of 2-4 years to present any robust determinations regarding success of the translocation. This is especially the case when the receptor site may fall within the effect zones associated with operation of the TLSB coastal lagoon development, for which the pilot study was conducted.

The translocated reef blocks were, however, in a healthy condition at the receptor site. This may indicate that the receptor site represents a more favourable environment for *S. alveolata* reef, than the site from which the reef was relocated. Further monitoring is required to validate any determinations of success or bias that may be present to increased environmental conditions at the receptor site.

The evidence provides a reasonably robust position that certain types of *S. alveolata* reef habitat can be translocated with an initial favourable result. It should also be noted that the evidence base is associated with a coastal lagoon development project located within Swansea Bay. Therefore it is relevant for consideration for other potential projects of similar type within Welsh waters.

The assessment presented above must also take account of potential changes in environmental conditions at the receptor site as a result of secondary environmental effects from the presence of the lagoon structure. As previously discussed, it is entirely possible/likely that alteration of hydrological and geomorphological conditions, including changes to nearbed sediment transport, may have an effect on survival/viability of a translocated reef at a receptor location. When screening the suitability of viable receptor locations, potential secondary effect pathways and footprints associated with the physical structure of the lagoon wall, and operation and maintenance effects should be taken into consideration.

5.3.6. Native Oyster beds

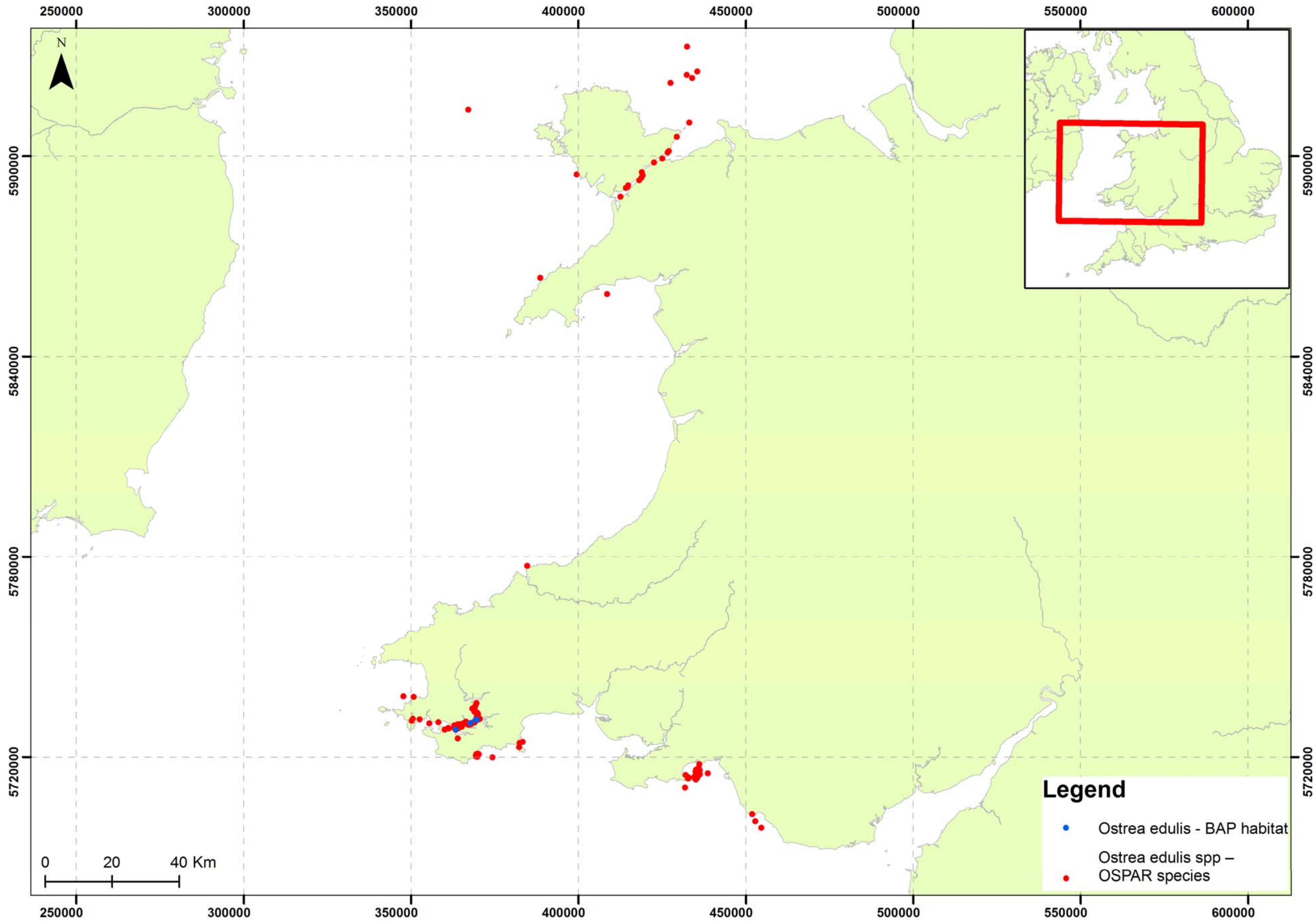
5.3.6.1. Introduction

The Native Oyster *Ostrea edulis* was once common in inshore and estuarine waters around the coast of the UK. However, from the late 19th to the early 20th Century, it underwent a significant decline across its entire geographic range. The Native Oyster population of the UK is characterised by an absence of recovery and continuing low abundance, and it is now subject to a UK BAP Species Action Plan.

The decline of Native Oyster and its failure to recover is attributed to a number of factors including fishing, habitat destruction, pollution, disease, and predation and competition from non-native species. The main UK stocks are now located in the rivers and flats bordering the Thames Estuary, the Solent, the River Fal, the west coast of Scotland and Lough Foyle, while some Native Oyster also remain in Swansea Bay. Woolmer *et al.* (2011) reported that this relic Welsh population, of generally older animals, is subject to some fishing pressure and to impacts from America Slipper Limpet *Crepidula fornicata*. Slipper Limpet is a non-native species that is a competitor of Native Oyster and can cause smothering at high densities. The Swansea Bay population is, however, clear of Bonamiasis, which is a notifiable disease of Native Oyster caused by the parasitic protist *Bonamia ostrea*. Bonamiasis is found widely in Native Oyster populations along the south coast of England, but is now also present in the Milford Haven population and at various other locations in Wales (Figure 5.3.4), Northern Ireland and Scotland (Laing *et al.*, 2014).

As part of the TLSB project, the reintroduction of Native Oyster to Swansea Bay has been proposed (TLSB ES Chapter 23; TLSB ES Appendix 8.3, TLSB, 2014b). Methods for restoring Native Oyster beds and of restoration efforts at other locations are discussed in the following sections in the context of the feasibility of such approaches at the TLSB site, and more generally in Wales.

Figure 5.3.4: Distribution of Native Oyster *Ostrea edulis* records in Welsh waters. (Source: NRW Marine Recorder snapshot 2015)



5.3.6.2. Approaches to restoration

Native Oysters have been studied for many years and much is known about their biology, ecology and distribution. As such, there is a considerable body of information on which to base restoration projects (Laing *et al.*, 2006). A very useful review of biological and physical information specific to the South Wales coast, including to Swansea Bay, was completed by Woolmer *et al*. (2011).

Efforts to restore Native Oyster beds may employ a variety of different methods that involve:

- Maintaining or enhancing seabed habitats to support spat[6] settlement;
- Creating or aggregating a local broodstock population;
- Collecting wild spat for seeding on to the seabed; and / or
- Restocking sites with hatchery-reared spat (Laing *et al.*, 2005; Laing *et al.*, 2006, Woolmer *et al*., 2011).

These approaches may be used individually or in-combination, depending on local factors and resource availability.

Maintaining or enhancing seabed habitats to support spat settlement is an essential starting point for Native Oyster restoration (Laing *et al*. 2005). Without appropriate habitat for settlement (i.e. cultch), any Native Oyster restoration project will be fundamentally compromised. In such circumstances it may require new animals to be sourced regularly through separate wild collection or by hatchery rearing, which is undesirable in the medium- to long-term. Natural production should therefore be encouraged instead.

On natural oyster beds, settlement surfaces include the shells of living and dead oysters, other shellfish and other hard substrata such as stones and wood. The growing edge of live oysters was assessed as the preferred settlement surface by Korringa (1946), but a low density of live oysters in most locations now means that alternative cultch must be provided. Old bivalve shells are often added as cultch in managed fisheries to encourage Native Oyster settlement (Laing *et al.*, 2005). Shells of different bivalve species are not universally suitable for settlement, however, and while oyster, scallop, slipper limpet and mussel shells seem to provide viable surfaces, cockle shells were reported to be less suitable (Key and Davidson, 1981).

When using shell cultch, flesh should be removed and shells exposed to the air for at least a month to minimise the risk of the spread of pathogens and non-native fauna from other areas (Bushek *et al*., 2004; Laing *et al*., 2005). Inert alternatives to shell such as crushed concrete or stones, may work effectively as cultch; although a study in Scotland found that Native Oyster spat preferred shell cultch (UMBS, 2007). The addition of non-natural materials may also be undesirable in a restoration project. The UMBS (2007) study also noted that the sparseness and patchiness of Native Oyster was likely to be exacerbated by the lack of suitable cultch.

Provision of cultch does not guarantee long-term viability of a Native Oyster bed, which can be subjected to periodic sediment deposition. A silt layer of 1-2 mm is reported to inhibit Native Oyster settlement (Laing *et al.*, 2005). Management on fished grounds therefore typically involves harrowing the cultch prior to the main spat settlement period, by running over the grounds with a

[6] Native Oyster larvae at the stage of settling to the seabed.

bagless oyster dredge or similar tool. This practice removes silt and exposes clean surfaces for the spat to settle on to. New cultch may also be needed periodically, as material is lost from the system through bad weather or because it sinks in to the sediment (e.g. Key and Davidson, 1981; Dumbauld *et al.*, 2000).

Powell and Klinck (2007) report that Eastern Oyster *Crassostrea virginica* abundance and shell production on fished oyster beds was not sustainable over the long term. They concluded that if fishing was permitted, or if disease had persistently increased the natural oyster mortality rate, then either shell cultch would have to be added in perpetuity to maintain the beds, or degradation of the shell beds would have to be accepted. This scenario may also be relevant to Native Oyster populations, although differences in species traits may obtain.

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. Where there are few adults, or where the adults are too dispersed, regular successful spawning may be impaired. Such situations may arise because of depensation effects where the sperm from male animals does not reach the eggs stored within the females. In these cases some element of broodstock enhancement may 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. Alternatively, when available, broodstock may be sourced from hatchery stocks. Both approaches have merit, but moving wild shellfish from one site to another increases the risk of introducing disease or non-native species to an area. Reliance on hatchery sourcing also increases the risk of limiting the genetic diversity of the population (Laing *et al*., 2005).

Wild collection is the main source of Native Oyster spat for commercial fisheries in a number of places, including for all French production (Oysterecover, 2013). A total of approximately 120 million spat are collected annually in France, with collection focused mainly in the Bay of Quiberon (90%) and with some collection also undertaken in the Bay of Brest (10%) (Oysterecover, 2013). Together with the presence of a relatively high abundance of adult animals, the somewhat enclosed nature of these two sites supports the collection of large quantities of Native Oyster spat from the wild. In more open systems, or where there is a minimal abundance of adults, the success of wild spat collection is likely to be much more variable. Evidence of this variability was found when different spat collection systems and media were tested at locations in France, the Netherlands and Denmark in 2011 and 2012 (Kamermans *et al*., 2013). Localised monitoring of wild spat production in Swansea Bay using a standard 'Chinese hat' spat collectors reportedly found no larvae in 2014 (MOC, 2014).

Techniques for cultivating Native Oyster spat in hatcheries or rearing ponds have been developed. Intensive commercial-level production is apparently undertaken in at least two UK locations (Ardtoe laboratory, Scotland, and Seasalter Shellfish, Whitstable). Although there are potential benefits from producing large quantities of spat in a short period, hatchery rearing of Native Oyster is not a simple or reliably successful process. Native Oyster are much less hardy than the Pacific Oyster *Crassostrea gigas* (Laing *et al.*, 2005), and production in hatcheries is reported to be erratic, with sudden and unexplained larval and post-metamorphosis mortalities (Oysterecover, 2013). As with sourcing a Native Oyster broodstock from hatchery stocks, maintaining genetic diversity is a key consideration for any restoration project utilising hatchery-produced spat. If hatcheries use a relatively small number of parents, the offspring will have low genetic diversity, with the subsequent potential for reduced fitness and impacts on wild populations (Launey *et al.*, 2001; Lapégue *et al.*, 2007).

Pond rearing of Native Oysters has the potential to address some of the genetic diversity issues associated with hatchery rearing. This is because, in comparison to hatcheries, a relatively large number of broodstock animals may be held together in ponds, such that a greater number of genetic crosses may occur at spawning (Lallias *et al.*, 2010). A Native Oyster pond production system has operated in Cork Harbour, Ireland, for a number of years. At that site, 21 ponds of approximately 1000 m^3 are each stocked with 700 –800 adults as broodstock (Oysterecover, 2013). An algal mix is cultured in the ponds and, after spawning, the spat are settled on to mussel shell, which are later scattered on to the seabed. After an on-growing period, the Irish system is reported to produce approximately 120 tonnes of Native Oysters, annually (Oysterecover, 2013).

5.3.6.3. External factors

A number of external factors will affect the potential for a Native Oyster restoration project to succeed in the long term. Critical factors are considered to be the use of appropriate fisheries management measures to regulate fishing activity on any newly established beds, and the prevention and management of diseases, predators and competitors of Native Oyster.

Fisheries management is therefore a key consideration in a Native Oyster restoration project. Fishing for Native Oysters is undertaken within a regulatory framework comprising national, regional and local management measures. Any individual may fish for native oysters in a public fishery so long as they do not contravene basic national measures and any additional local regulations. As such, and in the absence of specific measures to protect a recovering population, Native Oysters could be targeted and removed by fishermen operating legally and within the law. This issue was not addressed in the work undertaken to restore the Native Oyster population in Strangford Lough in the late 1990s; while that population initially benefited from the restoration efforts, unregulated harvesting reduced the stock again, probably to a level below the level where the fishery collapsed in the 19th Century (Roberts *et al*., 2005).

Disease and non-native species are important factors in the continuing poor status of UK Native Oyster populations (Laing *et al.*, 2014). Bonamisasis was first identified in Northern France in 1979, and can have devastating consequences for naïve populations (e.g., Laing et al (2014) calculated that French production of Native Oysters dropped by 93% between the 1970s and 1982, while Oysterecover (2013) reported that there was 98% Native Oyster mortality in the Cork Harbour rearing ponds when Bonamiasis first occurred there in 1987. Some resistance to Bonamiasis may be conferred on populations previously exposed to the disease as a result of spawning occurring from survivors (Beaumont *et al.*, 2005). However, populations may still suffer high morality and the initial avoidance of infection is a high priority, while infections may be managed through maintaining a relatively low density of animals on the beds (<10 m^{-2}), by minimising stress caused to the animals, and by harvesting at around age 3, after which the disease causes increasing levels of mortality. Harvesting may not be desirable, though, even if mortality may increase with age, if the aim of a restoration project is to enhance biodiversity rather than to support a fishery.

Other diseases of Native Oyster, include Marteiliosis, caused by the paramyxean parasite *Marteilia refringens*, and Denman Island Disease, caused by the protist *Mikrocytos mackini*. They are potentially very serious but do not currently occur within the UK (Cefas, 2014). American Tingle

Urosalpinx cinerea is also currently absent from Wales, but this non-native whelk species is a predator of Native Oysters and has the potential to devastate beds where it occurs (NNSS, 2014a). In contrast, Slipper Limpet is a non-native competitor of Native Oyster, and this species is now well-established around much of the south of the UK, including off the South Wales coastline. This species competes for food and space with other seabed species; Slipper Limpet can reach very high densities, at which point it can effectively smother and change seabed habitats (NNSS 2014b). Irrespective of whether some diseases or non-native species are already present in a site, however, careful consideration must be given to the source of any stock moved in to an area, and the movement controls and practices in place to prevent disease and non-native species transmission.

5.3.6.4. Confidence assessment

In order to assess the confidence in the evidence presented a confidence scoring system has been applied. The confidence matrix for Native Oyster is presented in Table 5.3.11 overleaf.

5.3.6.1. Summary

The evidence base is reasonably strong for the restoration of Native Oyster beds as a biodiversity enhancement measure, but weaker in relation to creation of the habitat itself. The confidence assessment assigns a low confidence. There are some comparable examples and the environmental parameters of functional beds are well understood. The evidence base and experience from temperate systems elsewhere in the world can be usefully referenced and drawn upon.

Correct identification of a suitable location within proximity to any proposed coastal lagoon development is clearly a fundamental factor. Water quality parameters and the high likelihood of maintenance dredging during the lifespan of these types of developments mean that successful and sustainable establishment of Native Oyster beds within the lagoon basin may be highly questionable. Full consideration of the sourcing of broodstock is equally important. Evidence shows that the production of viable spat is a complex process, requiring the use of established sources. This brings cost implications that have to be recognised if this habitat is to be seriously considered as a viable biodiversity enhancement measure.

An effective ES and AEMP will need to make full consideration and assessment of these likely constraints, in order to deliver a robust assessment to allow medium confidence that Native Oyster beds can be viable as a biodiversity enhancement measure; especially within a development’s lagoon basin footprint.

Table 5.3.11: Confidence assessment of viability of Native Oyster beds as biodiversity enhancement measure.

Keys

Information quality scoring system	1 = Unknown	2 = Inferred	3 = Known	4 = Well known	5 = Well known UK examples

Confidence level	**V. Low 1-2**	**Low: 2.1-3.5**	**Med: 3.6-4.5**	**High: >4.5**

Native Oyster Beds		
Factors considered	**Score**	**Notes**
Directly comparable projects for habitat	2	Restoration efforts in Strangford Lough (Roberts *et al.*, 2005) provide an indication of the approaches that may be taken for the TLSB project. However, that project focused on habitat and broodstock issues, not spat production as in the TLSB project. There was also work undertaken in Scotland, as reported by UMBS (2007), but again this did not focus on hatchery-based restoration.
Level of success	2	There are few commercial hatcheries for Native Oyster, associated concerns over the ability to make hatcheries work consistently well, and issues over genetic diversity. These are not clearly addressed in the TLSB proposal.
Environmental parameters of habitat understood	3	Native Oyster as a species is well studied, but there are variables that are not well known, including the risk of disease or harmful non-native species to Native Oysters in the Swansea Bay site, and the impact of climate change on Native Oyster and its susceptibility to disease.
Analogous examples	3	There has been good work undertaken in various research programmes in support of Native Oyster restoration (e.g., Blyth-Skyrme, 2011; Laing *et al.*, 2005; Oysterecover, 2013; SETTLE, 2012; UMBS, 2007; Woolmer *et al.*, 2011).
Temperate examples	2	There are a number of temperate oyster restoration programmes worldwide that may provide useful insight to the TLSB project team, including with Eastern Oyster (e.g., Powell and Klinck, 2007) and from New Zealand (e.g., Cranfield et al. 1999).
Tropical examples	N/A	
Total score	**12**	
No. of factors rated	**5**	

Average score	**2.4**	

Confidence level	**Low 2.4**

5.3.7. Artificial Substrata and reef effect

5.3.7.1. Introduction

Man-made structures that are introduced to the marine environment, such as breakwaters and seawalls, have the potential to increase the abundance and diversity of fish species found at particular locations. This effect is due to the structures acting as artificial reefs by increasing the complexity and diversity of available habitats.

Coastal lagoon development projects are likely to propose that habitats for intertidal and subtidal communities, and for non-migratory fish species, associated with the lagoon wall can be enhanced by using specific design approaches. Research on the effects of artificial reefs on marine communities and non-migratory fish is discussed below.

5.3.7.2. Approaches to restoration/enhancement

A large number of harbours walls, breakwaters and other structures have been introduced to the marine environment of the UK over recorded history. These structures are/have been designed primarily for purposes other than habitat creation or biodiversity enhancement. Artificial reefs have also been placed deliberately in many temperate locations in Europe over the last 40 years, including Finland, France, Portugal, Spain, The Netherlands and the UK (Jensen, 2002). The introduction of these hard structures into the marine environment can increase the diversity of local marine faunal and floral communities by introducing novel habitat types. The associated benefits may not always be positive, as such structures can facilitate establishment of invasive non-native species, and the loss of species associated with soft-sediment substrata (UNEP, 2009)(See Section 6). There is also some evidence that artificial reefs simply aggregate mobile species from surrounding waters rather than increase production or population size (Pickering and Whitmarsh, 1997; Polovina, 1989).

Inclusion of a large number and diversity of niche sizes in the substrate is a key component of artificial reef design for biodiversity enhancement (Browne and Chapman, 2011; Charbonnel *et al.*, 2002; Firth *et al*. 2013a, 2013b; Hunter and Sayer, 2009; UNEP, 2009). The use of relatively soft substrata has also been found to be important, as this provides a niche for boring animals (Moschella *et al.,* 2005).

Within the UK, six artificial reefs have been created since 1984, although only four of these have been designed specifically for habitat enhancement or associated research. These artificial reefs are located at Poole Bay and Salcombe in England and at Torness and Loch Linnhe in Scottish waters (Fabi *et al.*, 2011). The Poole Bay reef uses waste materials from power stations to create the reef as well as tyre modules. Monitoring of the reef has shown that epifaunal colonisation is rapid and provides a suitable habitat for Common Lobster *Homarus gammarus* and other commercial shellfish.

The Torness reef was constructed using quarried rock. Studies on the reef have shown that it has influenced local populations of Cod *Gadus morhua* that use the reef for refuge (Jensen, 2002). The

Loch Linnhe reef is also constructed from waste materials, utilising quarry dust slurry (Jensen, 2002). Research into suitable artificial material for use in artificial reefs construction was carried out at this site. The study examined the use of concrete, treated wood, rubber, steel and PVC. The results showed that the material used, and the season deployed, have an effect on the epifaunal assemblage in the short-term (three months) but these differences reduce as length of exposure increases. The study concluded that the material used will have little effect on the long-term community structure but the material should be physically stable, non-toxic and offer a high degree of habitat complexity (Brown, 2005). The Salcombe reef consists of natural rock (Fabi *et al.* 2011), but other published details do not appear to be available.

The substrate utilised in the construction of artificial habitats should be considered during planning and construction as previous research has shown that some species may settle preferentially on specific materials (Woodhead and Jacobson, 1985). However, a study has also shown that the amount, but not the type, of material used had a substantial influence on the fish assemblage at an artificial reef (Reed *et al.*, 2006).

According to the UNEP (2009) it is preferential to use natural materials in the construction of artificial reefs in European waters, although it is possible to use artificial materials providing that they are inert and resistant to deterioration in seawater. The use of concrete in artificial structures is compatible with the marine environment and is beneficial as it can be formed readily into any shape, which can provide adequate surfaces and habitats for the colonisation of organisms (UNEP, 2009; Pickering and Whitmarsh, 1997). The use of rock attracts fish and provides a good substrate for the settlement of organisms, and using a range of rock sizes can accommodate for different species and life stages. However, the type of rock used should be selected carefully as some rock types may contain heavy metals that may be released into the marine through leaching (UNEP, 2009). Although the material used should be considered carefully for each different project and location, the design of the reef structure may have more impact upon the success of colonising marine species than the construction material used (Pickering and Whitmarsh, 1997).

5.3.7.3. Benefits of restoration and enhancement

Artificial substrata in the marine environment provide a habitat for many species, although studies have shown that the level of biodiversity on artificial structures is lower than that found on natural habitats. The lower biodiversity may be due to the lack of habitat heterogeneity in artificial habitats, but the biodiversity in such habitats can be increased by providing rock pools which have been shown to support greater species richness at mid- and upper-tidal levels (Firth *et al.*, 2013a).

A number of ecological engineering options can be utilised to encourage the colonisation of new substrata, such as the drilling of rock pools into the structure. This approach has been used on the Tywyn breakwater in Wales, as well as the Plymouth breakwater in Devon, UK and was considered to be a low-cost option at both sites. The cost of drilling holes in the seawall at Tywyn was £860 and required two men working for two days. Another suggestion is the placement of gabions containing different sized rocks, which would provide a complex habitat as well as niches that would promote larval settlement. This was also considered to be a low-cost option valued as £500, although the size and number used is not stated. Gardening of artificial substrata has also been attempted at a site in Italy to improve colonisation, although this has only been attempted on a small scale. The cost of this was stated as being €6 per 15 x 15 cm. Therefore, it is possible that simple and cost-effective engineering methods can be employed in order to increase the colonisation and biodiversity on artificial structures (Firth *et al.*, 2013b).

A study of northern temperate artificial reef structures examined the species found on a gradient of increasing structural complexity (Hunter and Sayer, 2009). The results of this study found that the overall abundance for many of the species examined was 2-3 times higher on the complex artificial habitats than on simple artificial habitats or natural reefs. The study demonstrated that artificial reefs in northern temperate waters are capable of supporting assemblages that are at least equal in abundance and diversity to that of natural reefs. If the degree of complexity of the reef design is increased the enhanced habitat availability has the potential to promote species diversity and abundance above levels found on natural reefs. Therefore, artificial reefs may protect, enhance and augment populations of some species (Hunter and Sayer, 2009).

The geometry of artificial substratum has also been studied using plastic substrata placed in the intertidal zone. The use of such material allowed for the manipulation of structural and spatial features. The results of this study showed that the geometric characteristic that most influenced the epifaunal composition and density was the folding of the substratum, as this resulted in high faunal density and high initial colonisation. This proved to be more influential on the colonisation of epifauna than total area or volume of the substrate itself (Jacobi and Langevin, 1996).

There are a large number of artificial reefs within Europe (Jensen, 2002). Many of these artificial reefs have been shown to develop as successful ecosystems over prolonged periods of time, with five years having been shown to be a sufficient length of time for a relatively stable community to develop (Jensen *et al.*, 2000). Artificial reefs have been shown to increase fishery yields in reefs in Europe and have also been shown to provide suitable habitats for at least two species of lobster (Jensen *et al.*, 2000). Artificial reefs in Europe have also been used as protective devices for habitats, enforcing anti-trawling measures and protecting habitats such as seagrass beds (Jensen *et al.*, 2000).

A study of artificial reefs in Portugal examined the colonisation of sessile organisms (Boaventura *et al.*, 2006). The colonisation of sessile organisms on a reef structure will improve the food availability and shelter therefore making the reef more attractive to fish and other mobile species such as crabs. The study examined the colonisation of cubic concrete blocks after three and six months. After three months the species recorded were mostly barnacles, bryozoans and serpulid polychaetes. After six months a more diverse community was recorded including invertebrate groups such as porifera, hydrozoa, anthozoa, polychaeta, decapoda, gastropoda and bivalvia (Boaventura *et al.*, 2006). There is no information regarding comparisons with natural reefs in the area. However it is stated that artificial reef structures that are most similar in community structure to natural reefs are colonised by macroalgae, which was not recorded as present at the artificial reefs when the study was conducted.

The presence of artificial substrata in the marine environment has the potential to be colonised by local species but also by invasive non-native species. Research has suggested that invasive species may colonise or compete better on artificial substrata (Tyrell and Byers, 2007). A study examining invasive species of ascidians found that on natural substrata the occurrence of the invasive species was equal to that of native species. However, when communities developed on artificial substrata the abundance of invasive species increased and the abundance of native species decreased following initial colonisation (Tyrell and Byers, 2007). Therefore, the potential for colonisation by invasive non-native species should be a consideration when constructing an artificial structure. This is discussed in more depth in Section 6.

5.3.7.4. Artificial substrata habitat confidence assessment

In order to assess the confidence in the evidence presented a confidence scoring system has been applied. The confidence matrix for artificial substrata is presented in Table 5.3.12.

Table 5.3.12: Confidence assessment of viability of non-migratory fish habitat as biodiversity enhancement measure.

Keys

Information quality scoring system	1 = Unknown	2 = Inferred	3 = Known	4 = Well known	5 = Well known UK examples

Confidence level	**V. Low 1-2**	**Low: 2.1-3.5**	**Med: 3.6-4.5**	**High: >4.5**

Artificial Substrata Habitat		
Factors considered	**Score**	**Notes**
Directly comparable projects for habitat	3	There is a wide range of research that has looked at the colonisation of artificial reefs, including recent studies (e.g. Langhamer and Wilhelmsson, 2009; Jensen, 2002; Jensen *et al.*, 2002; Boaventura *et al.*, 2006; Hunter and Sayer, 2009). Nevertheless, there are few details regarding UK habitats or seawalls, so it is not possible to say that directly comparable projects for habitat are any more than 'known'.
Level of success	4	In the absence of detail on the design features of lagoon walls, it is not possible to know the potential level of success that may be achieved in enhancing communities. However, numerous approaches within the literature indicate strong success rates where artificial reef habitats are employed.
Environmental parameters of habitat understood	4	Details on the potential beneficial features that can be utilised within lagoon walls (use of other rock substrata, construction of rock pools, use and density of bioblocks, complexity of the rock surfaces, etc.) are known and can be used for coastal lagoon developments.
Analogous examples	3	Consideration of the colonisation of communities on artificial reefs and other structures within European and other temperate waters can be considered as analogous.
Temperate examples	4	A range of other artificial reefs have been constructed in Europe and in other temperate locations. There are also many thousands of piers or seawalls that have

		been constructed for harbours or as sea defences, all of which may provide useful examples of the benefits or otherwise for colonisation on the seawall.
Tropical examples	N/A	
Total score	**18**	
No. of factors rated	**5**	
Average score	**3.6**	

Confidence level	**Medium 3.6**

5.3.7.5. Non-migratory fish habitat

The design of the artificial reef is considered to be important if the reef habitat is to be successful and the target species should be considered. The reef should be constructed from a long-lasting material with plenty of niches (Jensen *et al.*, 2000). The effect of reef habitat complexity was investigated at a reef in France. The complexity of the artificial reef was increased by adding small-scale building materials into the chambers within the reef, and fish assemblages were monitored before and after the additions (Charbonnel *et al.*, 2002). The reef showed higher values of all community metrics measured after treatment, with species richness being twice as high, and biomass being forty times higher. Therefore, structural complexity plays an important role on diversity and abundance of fish on artificial reef structures (Charbonnel *et al.*, 2002).

Research has suggested that artificial structures designed for other purposes (e.g. breakwaters and jetties) could represent a potential tool for the population recovery and enhancement of fish species, especially if the area is protected from fishing activity. A study in the Adriatic Sea showed that the density of large- and medium-sized fish was greater at a protected breakwater compared with a breakwater where fishing was permitted. Therefore, artificial structures originally planned for other purposes may be beneficial to local fish populations so long as angling is restricted (Guidetti *et al.*, 2005). Other structures such as offshore renewable energy installations can also provide hard substrate for colonisation and act as a sanctuary area for certain fishery-targeted organisms. An expected outcome from such installations is higher survival of fish, increased individual size and maturation, possible increased fecundity related to larger age-classes with potential for spill-over into non-protected areas (Langhamer, 2012).

It is noted that the proposed restoration of Native Oyster beds (see Section 5.3.6) associated with coastal lagoon projects may be beneficial to fish species. Effects on fish and large crustacean by Oyster reefs in the southeast of the USA were studied by Peterson *et al.* (2003). They showed that an area of restored oyster reef 10 m^2 in size could be expected to yield an additional 2.6 kg yr^{-1} of production of fish and large mobile crustaceans for the functional lifetime of the reef. A reef lasting 20-30 years would be expected to augment fish and large mobile crustacean production by a cumulative amount of 38-50 kg 10 m^{-2} (Peterson *et al.*, 2003).

Non-reef structures such as offshore renewable devices have the potential to provide artificial substrata for colonisation. An experiment examined the foundations of wave energy foundations in Sweden as artificial reefs, additionally the potential for habitat enhancement was examined through manufactured holes in the foundations. A visual census of the foundations was conducted three months after deployment. The results found significantly higher numbers of fish and crabs on the foundations compared with the surrounding soft substrate. The holes within the foundations did not increase the abundance of fish, however, the increase of habitat complexity resulted in a five-fold increase in the densities of the Brown Crab *Cancer pagurus*. The density of the Spiny Starfish *Marthasterias glacialis* was shown to decrease due to complex habitats, although it is suggested that this may be a result of the increased predation by *C. pagurus* (Langhamer and Wilhelmsson, 2009).

The use of artificial reef structures to increase fish assemblages within an area has been criticised. There is the possibility that such structures may merely concentrate fish from surrounding dispersed areas at one location rather than increase the overall abundance of species within the region. This aggregation effect may result in increasing catch rates in areas where fishing occurs around the artificial structures. A review of the literatures by Grossman *et al.* (1997) found little evidence that artificial reefs increased regional fish production. However, another review of the available literature found evidence that the presence of artificial reefs increase fish production, although evidence is also presented for the opposing argument (Pickering and Whitmarsh, 1997). Ongoing discussions still occur regarding the 'attraction versus production' debate.

The scientific evidence and discussion about the potential benefits, limitations, and concerns associated with introducing hard structures to the marine environment, is well described in the TLSB ES in Appendix 8.3 (TLSB, 2014b). Rather than repeat the narration within this section **NRW staff are referred to Appendix A to this report where an extract of the relevant information from the TLSB Appendix 8.3 report is presented**. The potential role of bioblocks (1.5 m x 1.5 m x 1 m concrete structures designed to maximise habitat diversity and niche space) to the biodiversity enhancement strategy for the TLSB project is also discussed in Appendix 8.3 (TLSB, 2014b). No information is available regarding the optimal density of bioblocks that should be used to maximise diversity on an artificial structure. However, as discussed throughout this review, the greater the complexity of a structure the higher the levels of abundance and diversity found on the structures (Charbonnel *et al.*, 2002; Jacobi and Langevin, 1996). Therefore, it is reasonable to assume that a high proportion of bioblocks should be incorporated into the design of an artificial structure such as a tidal lagoon wall in order to increase the abundance and diversity of both the sessile communities, and the fish and crustacean species, that will colonise the structure.

5.3.7.6. Non-migratory fish Confidence assessment

In order to assess the confidence in the evidence presented a confidence scoring system has been applied. The confidence matrix for non-migratory fish is presented in Table 5.3.13 overleaf.

Table 5.3.13: Confidence assessment of viability of non-migratory fish habitat provision from artificial substrata as biodiversity enhancement measure.

Keys

Information quality scoring system	1 = Unknown	2 = Inferred	3 = Known	4 = Well known	5 = Well known UK examples

Confidence level	**V. Low 1-2**	**Low: 2.1-3.5**	**Med: 3.6-4.5**	**High: >4.5**

Non-migratory Fish Habitat		
Factors considered	**Score**	**Notes**
Directly comparable projects for habitat	3	There is a wide range of research that has looked at the effect of artificial reefs on fish and associated communities, including recent studies (e.g., Browne and Chapman, 2011; Firth *et al.*, 2013a, 2013b; Fowler and Booth, 2013; Koeck *et al.*, 2014). Nevertheless, there are few details on projects directly associated with seawalls.
Level of success	3	In the absence of detail on the design features of future lagoon walls, it is not possible to know the potential level of success that may be achieved in enhancing communities. However, approaches within the literature indicate success rates where artificial reef habitats can prove beneficial to fish species. However, it is not possible to know the potential level of success that may be achieved in enhancing non-migratory fish species.
Environmental parameters of habitat understood	3	Details on the potential beneficial features for fish species, that can be utilised within lagoon walls (complexity of the structure surfaces, etc.) are known, but may not be suitable for coastal lagoon developments.
Analogous examples	2	Suitable analogues can only be inferred. Most examples are associated with complex subtidal structures.
Temperate examples	3	A range of artificial reefs have been constructed in Europe and in other temperate locations. There are also many thousands of piers or seawalls that have been constructed for harbours or as sea defences, all of which may provide useful examples of the benefits/dis-benefits of the lagoon walls as habitat for non-migratory fish species.
Tropical examples	N/A	
Total score	**14**	

No. of factors rated	**5**	
Average score	**2.8**	

Confidence level	**Low 2.8**

5.3.7.7. Summary

A wide variety of studies of the effects of artificial reefs have been published over the last two decades, and the review undertaken for the ES Appendix 8.3 is appropriate (see Appendix A). Evidence regarding the biodiversity benefits of seawalls for intertidal and subtidal flora and fauna is well-documented, and such structures are analogous to coastal lagoon development walls. There is a medium confidence that habitat enhancement is achievable, where hard substrata reef communities are deemed acceptable.

Delivery of enhanced habitat, from lagoon wall artificial reef effects for estuarine non-migratory fish is not as robust as for offshore examples of artificial reefs. Accordingly there is a low level of confidence in achieving biodiversity enhancement for estuarine non-migratory fish through the appropriate use of known/current methods.

The issue of lagoon wall design should clearly be considered for any future coastal lagoon development projects, especially regarding the use of bioblocks and other biodiversity enhancing measures such as rockpool creation, sympathetic sourcing of local rocks for gabions and general high structural complexity. The use of reef enhancing mechanisms and methods need to be fully considered with NRW staff, and adhered to throughout the entirety of the application process and ‘locked in’ to a project. Without these considerations making it through the process and into actual construction, then any positive effects will likely not be realised.

Considering the low technology expenditure in achieving a potential positive biodiversity enhancement measure, this topic is one that should be highlighted and stipulated much more clearly in any statutory advice. The potential for easy wins, at least to off-set some of the likely environmental impacts of such developments means that delivery of reef enhancement measures should be pursued.

5.3.8. Atlantic Herring spawning bed habitat

5.3.8.1. Introduction

Atlantic Herring *Clupea harengus* is a pelagic fish that is widespread in UK waters. The spawning activities of Atlantic Herring include the deposition of demersal eggs to a suitable attachment surface where they remain until the larvae have developed sufficiently to join the plankton; at approximately 6-11 mm length. Sediment habitat with the potential to support spawning beds have been found to include Gravel through sandy Gravel to gravelly Sand based on the Folk (1954) sediment classification (Reach *et al.,* 2013).

They are several distinct spawning components of Atlantic Herring known within UK waters (ICES, 2014). The predominant components of the spawning population within the North Sea are the

Orkney/Shetland, Buchan, Banks and Downs (MarineSpace *et al.*, 2013; Reach *et al.*, 2013; ICES, 2014). The Downs population also extends into the east English Channel and along the south coast of England to around the Isle of Wight. All of these spawning components are referred to as **North Sea Autumn Spawners** (NSAS). Spawning begins around September/October in Scottish waters, and then progresses southwards through the late autumn and early winter. The Downs population spawns around the end of November through to mid-January (ICES, 2014).

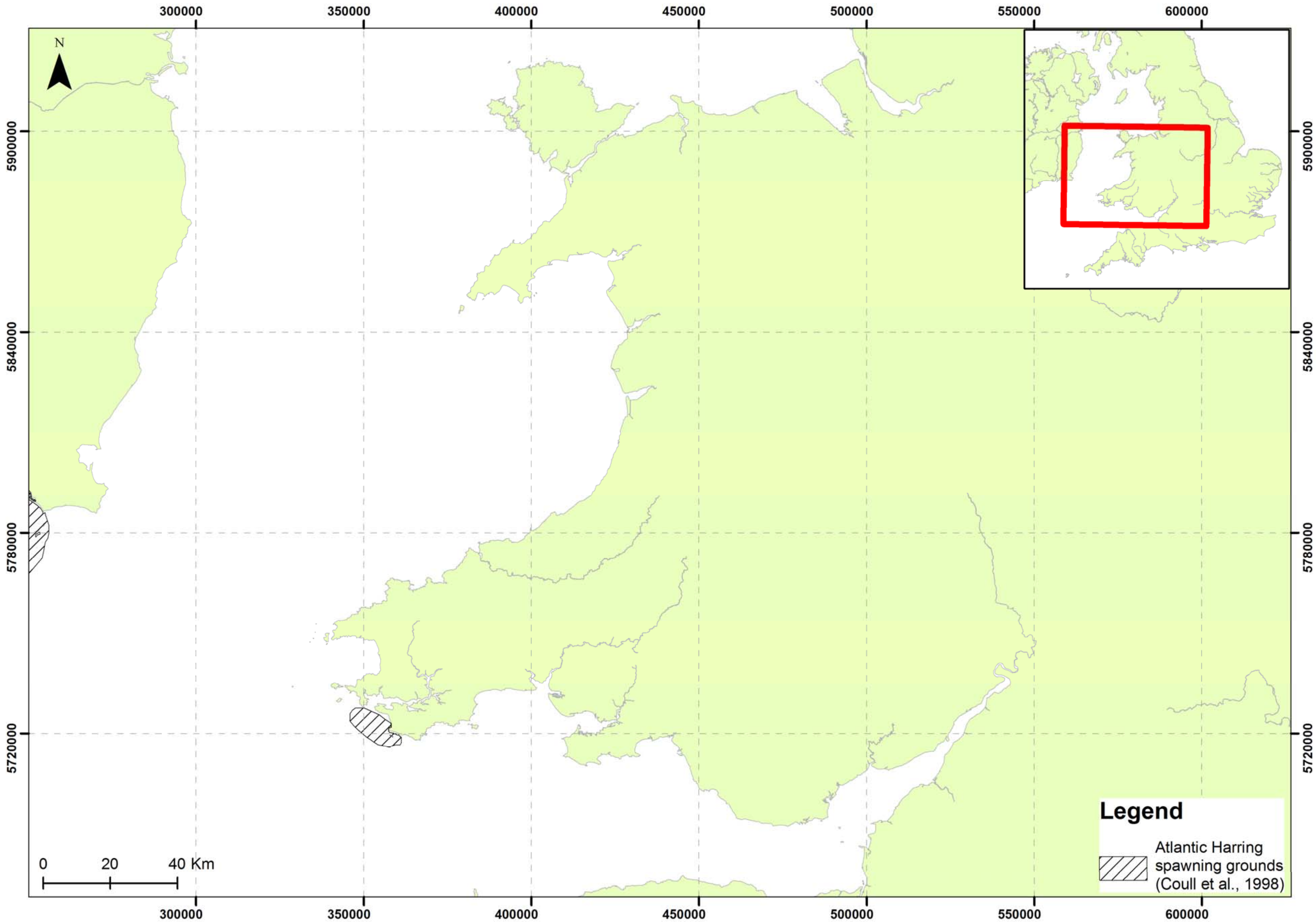
Along with the major autumn spawning populations, there are certain discrete sub-populations of Atlantic Herring that are associated with spawning at large estuaries and embayments around the UK. Most notable are spawning aggregations associated with the mouth of The Wash, the Blackwater Estuary within the outer Thames Estuary, and The Severn Estuary (Ellis *et al.*, 2012). These estuarine sub-populations are referred to as **Estuarine Spring Spawners** (ESS), with spawning events variable through February-May (Titmus *et al.*, 1978; ES Section 9, TLSB, 2014b; TLSB 9.1, TLSB, 2014b).

In general, discrete spawning beds associated with ESS are poorly understood or detailed within national and international data sets (Coull *et al.*, 1998; Ellis, 2012; ICES, 2013) (Figure 5.3.5). It is apparent that the most relevant information about local sub-populations of ESS resides with local fishers. Research conducted by O'Sullivan *et al.* (2013) demonstrated that the use of local fisher's marks was a reliable way of mapping and assessing spawning beds for Atlantic Herring in Irish waters. Development of further lagoon development projects should therefore include dialogue with local fishers, and relevant representative bodies/organisations, to provide the most relevant resolution of data required for EIA at a project-scale.

As identified within the TLSB ES, there will be impact upon known spawning beds within Swansea Bay (TLSB 9.1, TLSB, 2014b). This indicates that similar effect-receptor pathways may exist for future coastal lagoon development proposals. The feasibility of creating Atlantic Herring spawning grounds as an enhancement measure to offset direct impacts such as substratum loss/habitat removal associated with constructing lagoon walls is clearly of interest to NRW. A literature search has not produced any results for the creation of Atlantic Herring spawning habitat anywhere in the World, aside from those described within the TLSB ES and supplementary documents (TLSB 9.1, TLSB, 2014b). The latter enhancement measures are discussed later in this sub-section.

Relevant environmental parameters for preferred potential spawning seabed habitat for NSAS Atlantic Herring are discussed below. Requirements of ESS appear much more variable, restricted and dependent on localised features and environmental parameters, than for NSAS.

Figure 5.3.5: Recorded Atlantic Herring *Clupea harengus* spawning grounds in Welsh waters. (Source: Coull *et al.*, 1998)



5.3.8.2. Environmental parameters

Recent investigations by Reach *et al.* (2013) show that Atlantic Herring spawning beds for NSAS may include seabed sediments of Gravel through sandy Gravel to gravelly Sand (as determined by Folk, 1954). Whilst these Atlantic Herring potential spawning seabed habitats appear to have a relatively wide range of distribution, spawning events of NSAS are spatially variable and in any year restricted to discrete spawning beds found at a local scale (MarineSpace *et al.*, 2013; O'Sullivan *et al.*, 2013; ICES, 2014). Whilst preferred sediment types within known NSAS spawning grounds are widespread other broader environmental characteristics and parameters are also important, such as: oxygenation of sediments; near bed flow rates; and micro-scale seabed morphological features e.g. ripples and ridges (de Groot, 1979, 1980, 1986, 1996; Bowers, 1980; Rankine, 1986; Aneer, 1989; Blaxter, 1990; Morrison *et al.*, 1991; Heath *et al.*, 1997; Maravellias *et al.*, 2000; Maravellias, 2001; Mills *et al.*, 2003; Skaret *et al.*, 2003; Geffen, 2009; Payne, 2010; ECA and RPS Energy, 2010a, 2010b, 2011; ICES, 2012).

Considering the variable environmental parameters it is difficult to map the nature of these seabed areas utilised for spawning at a fine-scale (MarineSpace Ltd *et al*., 2013). This is further complicated as spawning beds may be used sporadically, laying unused for several years at a time, before being re-used (Bowers, 1980; Maravellias, 2001; Skaret *et al.*, 2003; Schmidt *et al.*, 2009; Corten, 2013).

As well as substrate, Atlantic Herring spawning areas are also influenced by water depth. Spawning of Atlantic Herring in Scotland generally takes place in depths of 10 m or less, however deeper spawning beds have been observed (Aneer, 1989). A study of Atlantic Herring spawning beds off the coast of Ireland found that spawning beds ranged from depths of 7-90 m. In the Celtic Sea the average depth of spawning beds was approximately 30 m and were located no more than 2 miles offshore usually within sheltered bays and estuaries (O'Sullivan *et al.*, 2013).

Siltation of NSAS spawning beds is a major impact pathway, resulting in smothering of eggs and yolk-sac larvae, with associated mortality (de Groot, 1980, 1986; Aneer, 1989; Morrison *et al.*, 1991; Geffen, 2009; ICES, 2014). For ESS suspended sediment concentrations may not be so limiting, however it should be noted that tidal flow rates associated with hard substrata, including seawalls, may enable winnowing of fine sediments naturally mitigating smothering effects (this is a hypothesis with no evidence to support the statement).

In relation to ESS, which are likely to be most relevant to Welsh coastal lagoon development proposals, it is apparent that further environmental parameters may be important for discrete spawning beds. TLSB 9.1 (TLSB, 2014b) indicates that along with coarse sediments providing preferred seabed spawning habitat, there are records of spawning events associated with hard substrata such as bedrock, boulders, *Sabellaria alveolata* reefs, and artificial structures such as sea defences.

5.3.8.3. Approaches to restoration/enhancement

In the case of ESS which use hard substrata for spawning beds, enhancement measures may reasonably consider the use of hard substrata, including possibly lagoon infrastructure itself, for habitat provision (as proposed with TLSB 9.1, TLSB, 2014b). It is critical to note, however, that there is an extremely limited evidence base concerning artificial habitat creation delivered for Atlantic Herring anywhere in the World, especially ESS (de Groot, 1980). Another example of herring spawning associated with artificial structures in an area known to support previous spawning beds has been recorded (U.S. Army Corps of Engineers, 2013). A precautionary approach to this evidence is advised as it is associated with Pacific Herring *Clupea pallasii*. Unlike Atlantic Herring, Pacific Herring demonstrates much higher affinities to hard substrata spawning, although this may be more analogous for ESS Atlantic Herring (Lassuy, 1989). In these cases, spawning has been recorded upon hard substrata replacing existing areas of spawning substrata. Overall success is difficult to assess considering the known inter-annual variation of spawning bed fidelity and variation in environmental parameters such as temperature and salinity fluctuation, stock status and recruitment success (ICES, 2014).

Replacing existing similar spawning beds with new, suitable, hard habitat is potentially viable habitat enhancement. There are, however, no data available indicative of responses from discrete ESS populations in response to translocation of areas of preferred spawning habitat (i.e. there is no demonstrable evidence base that such populations will relocate to a different area just because favourable seabed habitat has been created away from historical spawning beds). Extreme care should therefore to be exercised if considering spawning bed translocation for habitat enhancement or mitigation.

5.3.8.4. Confidence assessment

In order to assess the confidence in the evidence a confidence table is presented. The confidence table for Atlantic Herring spawning bed habitat is presented in Table 5.3.14 overleaf.

Table 5.3.14: Confidence assessment of viability of Atlantic Herring spawning bed habitat as a biodiversity enhancement measure.

Keys

Information quality scoring system	1 = Unknown	2 = Inferred	3 = Known	4 = Well known	5 = Well known UK examples

Confidence level	**V. Low 1-2**	**Low: 2.1-3.5**	**Med: 3.6-4.5**	**High: >4.5**

Atlantic Herring spawning bed habitat		
Factors considered	**Score**	**Notes**
Directly comparable projects for habitat	2	There is only one example of creation of Atlantic Herring spawning bed habitat available within the literature review.
Level of success	2	Two examples assessed showed a degree of success although the time-series monitoring data is limited. Therefore confidence in success is likewise limited.
Environmental parameters of habitat understood	2	Whilst the use of seabed substrata to support spawning beds for Estuarine Spring Spawner Atlantic Herring is known, the variation of other critical environmental parameters is little understood.
Analogous examples	2	There is only one example in the literature base for herring spawning bed habitat creation. That relates to the Pacific Herring and spawning on a seawall. Considering the similarities in spawning behaviour between Pacific Herring and Estuarine Spring Spawner Atlantic Herring within the Severn Estuary the analogy is determined relevant.
Temperate examples	3	Both known examples of habitat creation are from temperate systems: one in the Netherlands, and the second from Alaska (Arctic-Boreal).
Tropical examples	1	No temperate examples were found.
Total score	**12**	
No. of factors rated	**6**	
Average score	**2.0**	

Confidence level	**Very Low 2.0**

5.3.8.5. Summary

The evidence base is weak for the creation of Atlantic Herring spawning bed habitat for Estuarine Spring Spawners (ESS) as a biodiversity enhancement or mitigation measure. There is no actual evidence base for translocation of the spawning bed habitat and should effectively not be considered possible. There is also no evidence to support suppositions that ESS displaced from discrete historical spawning beds will simply relocate to the next available preferred seabed habitat. Within the time span of potential new coastal lagoon development projects there can also be no confidence in habitat translocation.

The two examples reviewed do indicate a level of success associated with habitat creation/ replacement, but it is apparent that such an enhancement measure should, at this stage of the knowledge base, be considered a trial or pilot study. It is doubtful that the methodology can be considered an effective enhancement measure at this time, as the confidence is very low and risk of failure considered medium-high. Implementation of pilot studies associated with any project may be advisable, to validate the efficacy of the approach. However, considering the high degree of environmental variables associated with the location of established spawning beds, any robust scientific determinations may difficult to attain. Successful spawning events, could however be considered the only result required, certainly if demonstrable beyond a sing annual event.

As has been discussed the known parameters for spawning bed habitat are complex and include variable parameters such as oxygenation of seabed, sediment loads, flow rates, wave exposure and suitability of seabed substrate. This makes generic statements regarding habitat creation difficult with site-specific investigations/pilot studies preceding construction a fundamental requirement. Without this pre-construction validation the creation of Atlantic Herring spawning bed habitat may not be considered a viable enhancement measure.

The confidence assessment therefore assigns a very low confidence score for habitat creation. There is only a single example of spawning bed habitat creation/replacement for the species, known in the world, and that has only time-limited amount of monitoring data available (de Groot, 1980). A further example is available (used as part of the TLSB ES and supplementary documentation, TLSB 9.1, TLSB, 2014b), but this relates an example for Pacific Herring, not Atlantic Herring (U.S. Army Corps of Engineers, 2013).

6. Invasive Non-native Species and Biosecurity

6.1. Introduction

Invasive non-native species are described as "*organisms introduced by man into places outside of their natural range of distribution, where they become established and disperse, generating a negative impact on the local ecosystem and species*" (IUCN, 2011). There are principal vectors for the successful establishment of marine invasive non-native species (MINNS). These comprise biofouling (unwanted attachment of fouling organisms on ship hulls and man-made structures), ballast water transport of viable plankton stage larvae (Bax *et al*., 2003) and offshore and nearshore/inshore 'stepping stone' habitats such as oil platforms and renewable energy structures (Adams *et al*., 2014), marina and coastal defences which providing artificial habitat facilitating colonisation by MINNS (Bulleri and Chapman, 2010).

As noted by Diamond (1989) invasive species were implicated as one of the four major reasons for global biodiversity decline. Accordingly, invasive non-natives are now recognised as one of the leading causes of biodiversity loss and in marine ecosystems with drives for global trade and new shipping routes opening, this problem may be increasing (Natural England, 2009; Seebens *et al*., 2013). The United Kingdom is a major recipient of MINNS through biofouling/ballast water in addition to accidental introduction through aquaculture transfer (Lambert, 2009).

6.2. Example species and related effects overview

In Welsh waters, a number of MINNS are already noted. Focus species, including some of the most invasive and problematic in UK waters, were identified by the Wales Working Group on Invasive Non-native Species in two documents published in 2010 (Non Native Species Secretariat, 2015a). This information, in conjunction with recent data (Priority List) supplied by Natural Resources Wales, has been used to create pertinent example MINNS information given below. These data were considered in addition with species with 'Alarm' status and an 'action audit' list found on the Wales Biodiversity Partnership –Invasive Non-Native Species Group website (BiodiversityWales, 2015).

Not all species that arrive from areas outside the UK may be considered invasive. An invasive organism is one that is wittingly, or unwittingly, introduced by man into places out of their natural range, and becomes established generating a negative economic (e.g. ecosystem service) or environmental (ecological) impact on the local, and possibly wider, ecosystem and species (see Morris and Whitfield, 2009). Notable examples considered through wider UK (Non Native Species Secretariat), the Wales Biodiversity Partnership and Natural Resources Wales, include:

- The **American Slipper Limpet *Crepidula fornicata***, a widely recognised invasive gastropod that arrived in the UK (Liverpool Bay) in the 1870s, died out, then was reintroduced to Essex in the late 1800s from where the population spread occurred round UK coasts (see Minchin *et al*., 1995). Accidentally introduced with Eastern (American) Oysters *Crassostrea virginica* and the American Hard Shell C*lam Mercenaria mercenaria* and also considered as transportable by slow-moving hulls (Mineur *et al.*, 2012), and possibly in ballast water as larvae (JNCCa, 2006). It alters habitat through physical numbers and through faecal material and pseudofaeces (Katsanevakis *et al.*, 2014) creating organic rich fines that create anoxic areas and may impede filtering species *(*de Montaudouin *et al*., 1999). Thus, *C. fornicata* can impact habitat for indigenous Native Oyster *Ostrea edulis* and imported Eastern Oyster *Crassostrea virginica* species with which it first arrived in the UK. *C. fornicata* is in the Welsh invasive species priority list having been in south Welsh waters since 1953, and having arrived in North Wales (Menai and Conwy Bay SAC) via contaminated mussel transfer (though being controlled by removal and smothering). Its effects on commercial shellfisheries through substrate alteration and physical impingement of harvesting (Payne *et al.*, 2014) represent a significant ecosystem service effect through food reduction and financial impact. Noting in the Welsh Priority list that local stakeholders may "*choose to manage*", a wider suggestion is made by Bohn *et al.* (2013) that:

"*…understanding recruitment patterns in non-native species is essential for developing management strategies for potentially harmful invaders such as* C. fornicata*.*";

- In similar fashion to *C. fornicata* the invasive **Carpet Sea Squirt *Didemnum vexillum*** and the **Leathery Sea Squirt *Styela clava*** both have implications for aquaculture industries through smothering, with subsequent implication for food supply and finances through management works and control of transport vectors (Payne *et al*., 2014). In addition *D. vexillum* has been suggested to have negative effects on benthic habitats and suggested keystone species through smothering and alteration of CO_2 budget (see Dupont *et al*., 2007; Gittenberger, 2008). For the UK *D. vexillum* was first recorded in Holyhead harbour (North Wales) and latterly on the UK south coast; transport through biofouling or ballast water was implicated (Griffith *et al*., 2009). For *S. clava* the transport vector was fouled ships arriving in the UK in the early 1950s. Noted on the Welsh priority list, action was undertaken in Holyhead Marina to kill *D. vexillum* at a cost of some £800K (Payne *et al*., 2014); monitoring actions continue. For *S. clava* localised efforts to modify abiotic factors (emersion, salinity etc.) have proven effective (JNCCb, 2006) although for both species, monitoring of vessels and adherence to ballast water and biofouling management plans are the preferable proactive routes;

- Described as "*locally invasive*" in the Welsh priority species list, with Strategic Control Priority status, **Devil's Tongue Weed *Grateloupia turuturu*** is an Asian invasive algae likely to arrive on either contaminated shellfish, by hull transport or possibly through contaminated ballast water (GB Non-Native Factsheet, 2011). First recorded on the UK south coast, it was noted in Milford Haven in 1984. Whilst implications for ecology and/or ecosystem services with benefit to humans are little researched, some work has shown that *G. turuturu* can smother indigenous algal species resulting in impacts on trophic food webs with loss of resource for grazing gastropods. In addition it has been suggested that the physical size of the algae can also impact fishing gear and shellfish beds and interfere with harvesting (Katsanevakis *et al.*, 2014); and

- The Alarm List (see below) contains details of species:

"*…thought to pose a risk to surface waters and their ecological status under the EC Water Framework Directive, but whose presence has not yet been recorded in Great Britain*".

> As an example, of notable species within the list, the **Japanese Oyster Drill *Ocinebrellus inornatus*** which can be confused with the European Sting Winkle *O. erinacea*, is a pest. This has been transported through contaminated shellfish species and may be a significant threat for indigenous Native Oyster, having already been shown to impact beds in France and to be a possible threat in Denmark (Lutzen *et al*., 2011).

The above are a limited list of examples of non-indigenous marine species with proven or potential ability to impact ecosystems or ecosystem services. As initial broad resources, further information on the numerous MINNS of concern in Wales can be viewed at:

- http://www.nonnativespecies.org/index.cfm?pageid=173;
- https://secure.fera.defra.gov.uk/nonnativespecies/downloadDocument.cfm?id=249; https://secure.fera.defra.gov.uk/nonnativespecies/downloadDocument.cfm?id=248; and
- http://www.biodiversitywales.org.uk/42/en-GB/Invasive-Non-Native-Group where the *Alarm List* can be downloaded.

6.3. Legislation, regulation and guidance

There are numerous legislation instruments and guidelines to control for, or indicate the significance of, impacts and effects from non-native species. The UK is a signatory to the Convention on Biological Diversity (CBD) and Aichi Target 9 indicates that biosecurity planning may become mandatory at a project design phase thus this has driven a more unified EU approach to Invasive Alien Species (IAS) formerly lacking in Europe. This came into place on 01 January 2015 and is known as the Invasive Alien Species Regulation (REGULATION (EU) No 1143/2014[7], agreed October 2014. The regulation advises three types of intervention pertinent to invasive species control being "*prevention, early warning and rapid response, and management*" (Non Native Species Secretariat, 2015b). However, it is not the purpose of this document to provide exhaustive overview of pertinent legislation and regulation, but a short list of important items is given here:

- EU IAS Regulation (1st January 2015), providing a unified EU approach on the proactive and reactive management of invasive species (see: http://www.nonnativespecies.org/index.cfm?sectionid=78);
- The UK is party to other EU conventions and legislation (see: http://www.nonnativespecies.org/index.cfm?sectionid=83), including the Water Framework Directive (A framework to achieve Good Ecological Status in "*transitional and coastal waters by 2015*[8]"); and this relates to:
- The Marine Strategy Framework Directive (MSFD) which requires that signatory states use the ecosystem approach (ecosystem services) to achieve "*good environmental status*" by 2021. Notably item 2 of the MSFD states that "*Non indigenous species introduced by human activities are at levels that do not adversely alter the ecosystems*" (see: http://jncc.defra.gov.uk/page-5231#GES1);

[7] see: http://www.nonnativespecies.org/index.cfm?pageid=211
[8] This date may change as river basin management plans are currently under consultation (relating to WFD) to extend to 2021.

- England and Wales legislation can be viewed at: http://www.nonnativespecies.org/index.cfm?pageid=67;
- The International Maritime Organisation (IMO) Convention for the Control and Management of Ships' Ballast Water and Sediments (BWMC) was adopted in 2004 and if ratified by 30 states representing 35 percent of the world's merchant shipping tonnage will enter into force 12 months later. Details of the BWMC and UK guidelines for ballast water management can be found here: https://www.gov.uk/control-and-management-of-ballast-water; and
- The IMO has produced the Biofouling Guidelines aimed at vessel and vessel owner responsibility for ensuring a ship or structure underwater area is managed and free of fouling species. The Guidelines can be viewed here: http://www.imo.org/blast/blastDataHelper.asp?data_id=30766&filename=207%2862%29.pdf. Whilst the Guidelines do not stipulate, awareness of underwater cleaning methods and potential pollution events in relation to local and national or EU guidelines should be considered.

6.4. Tidal lagoon structure design and marine invasive non-native species

Proposals for tidal lagoon developments should consider the appropriate mitigation and control measures in addition to assessing the risks posed by known MINNS. In the case of invasive species, prevention is better than cure and if no attempt to manage the risk is undertaken, this may have significant direct or cumulative financial and/or ecological consequences. The introduction of a hard structure (such as a lagoon wall) could provide a 'stepping stone' for both indigenous and non-native species that results in epibiota that differ from the surrounding natural environment (Mineur *et al.*, 2012).

Man-made intertidal/sub tidal structures can create novel habitat for biota that favour a hard substrate where otherwise there is none, and can encourage the colonisation and spread of MINNS (Bulleri and Chapman, 2010). Research has shown that as the presence and extent of intertidal structures increase, opportunities for both indigenous and exotic species also increase, potentially leading to invasion of a natural habitat by an invasive species (Bulleri *et al.*, 2006). Relevant structures include coastal flood defences, harbour or general infrastructure, and theoretically a lagoon wall.

MINNS have been the subject of a considerable amount of research. Lagoon development projects should therefore ensure that a thorough review of such information is presented in any project's ES. This will assist decisions regarding intertidal/subtidal substrate and structure design, such that ecological and invasive species aspects are considered. It will also satisfy EU objectives for establishing and using proactive approaches to minimise risk from MINNS.

Current research shows that proliferation of man-made structures can favour exotic species. This is because MINNS are competitive in disturbed, low diversity environments. In a geographical region where evolution has not presented a natural predator-prey relationship, and no natural stop points are present, MINNS are able to dominate these types of environments. Man-made structures can require frequent maintenance or modification. This disturbance prevents development of a robust diverse indigenous species assemblage and effectively creates an early succession community favouring MINNS (Glasby *et al*., 2007); higher diversity is better able to buffer or rebuff the effects of invasive species, potentially related to resource (space) availability (Stachowicz *et al.*, 2002). Further to the ability to outcompete in low diversity communities, invasive non-native species may also be more competitive over a wider range of substrates than native species, and this may result in increased numbers of non-natives on artificial structures (Tyrrell and Byers, 2007; Glasby *et al.*, 2007).

It is widely recognised that a more diverse habitat will generally lead to a diverse community better able to withstand invasive species. Enhancement of natural biodiversity associated with the project footprint assists biosecurity. High biodiversity has been shown to increase the resistance of a community to invasion from non-native species. Lagoon wall biodiversity may be enriched through the use of structures such as bioblocks, or reef balls, and 'seeding' them with native communities from adjacent areas.

Recommendations made at the design stage which can be considered as working with biodiversity are likely to enhance structures. Airoldi *et al*. (2005) reported on the DELOS (Environmental Design of Low Crested Coastal Defence Structures) project and indicated a series of design factors that may be applicable in limiting success of MINNS. It is therefore recommended that at the final design phase, there should be detailed consultation with relevant experts to assess the risks associated with:

- Substrate;
- Wall design;
- Source of construction heavy plant and associated infrastructure, including jack-up rigs etc.;
- Maintenance schedules/disturbance aspects; and
- Linkages with other structures potentially facilitating the 'stepping stone' effect.

All require detailed consideration to meet EU and Welsh goals of pro-active management of invasive species. Further, appropriate monitoring (see below) should be employed through post-project funding to enable early consideration of invasive species and management responses.

6.5. Mitigating introduction or spread of invasive non-native species

Primary vectors for MINNS generally comprise:

- Biofouling on incoming commercial including construction and support vessels, and recreational vessels;
- Contaminated ballast water;
- Transfer of fouled man-made structures;
- Movement of contaminated shellfishery species; and
- The facilitation of these species by creation of 'stepping stone' habitat.

Mitigation of these factors, where feasible, should be undertaken with the new EU and Welsh goals of minimising invasive species colonisation.

Prior to construction, mitigation measures should be established to address issues posed by potential vectors and facilitating factors for MINNS. Long-standing research has clearly identified the role that shipping has spreading MINNS. Signatories to the Ballast Water Convention, and latterly adoption of the Biofouling Guidelines, indicate that there is an international will to manage the issue. However, it's also recognised that commercial shipping is easier to regulate and manage than small recreational and localised commercial boats. These smaller vessels result in almost unmanageable factors at the vessel level, thus the harbours or marinas in which they reside increasingly need to show precautionary management of invasive species.

Apart from the direct or indirect involvement of craft or structures, the other main vector is transplant of infected shellfish. There is a substantial risk that invasive species will propagate and potentially utilise man-made hard substrate (concrete is known to be a good facilitator of invasive spread (Glasby *et al*., 2007)). This is an important consideration where there are proposals to establish Native Oyster beds.

Finally, transplant of invasive species already existing at a site is an important consideration requiring pre-survey, or robust characterisation baseline data to ensure informed action before construction phases.

Given the range of issues associated with MINNS, a biosecurity risk assessment is an imperative. Guidelines are available at http://www.snh.gov.uk/docs/A1294630.pdf (Payne *et al*., 2014). Mitigation measures within a biosecurity risk assessment should include, but not be limited to:

- Seek advice from NRW and relevant parties for information on extant MINNS populations;
- Put in place an identification and reporting procedure prior to any survey;
- Preliminary survey of area with assessment of MINNS and consideration of possible enhancement of their populations through industrial activity (e.g. dredging movement of *C. fornicata* to new locations – implications under the Wildlife and Countryside Act, Variation No.7609, (2010)[9];
- Ballast water management for construction vessels (IMO Guidelines);
- Hull fouling management and assessment for construction and attending vessels (IMO Guidelines);
- Particular consideration of vessels or structures being moved at less than 4 knots where biofouling can occur in transit as well as when stationary (Chambers *et al.*, 2006);
- Locality of any shellfisheries and implications for transport both from and to of MINNS;
- Design of structures to include substrate, e.g. limestone and appropriate numbers of bioblocks/habitat that encourages settlement and colonisation of indigenous species (see 5.3.5.2) through provision of microhabitat, rugosity and suitable substrate; and
- Pre- and post-construction monitoring for agreed periods, to establish baseline structure habitat and colonisation of indigenous/non indigenous species.

6.6. Prevention

It is widely accepted that prevention is preferable to cure. This is especially true in the case of MINNS as marine systems are relatively 'open' thus eradication post-arrival is financially and

[9] See: http://www.legislation.gov.uk/uksi/2010/609/pdfs/uksiem_20100609_en.pdf)

logistically challenging. Research has shown that natural, healthy communities can offer the best resistance to MINNS. Therefore the seeding of newly constructed seawalls with propagules, larvae, spat, or through translocation of mature individuals, or limiting the disturbance to surrounding habitats during construction, can increase the potential for resistance to their colonisation.

However, the most effective prevention is to provide clear guidelines based on legislation and EU/Welsh/UK goals with regard to managing the vectors for MINNS. This constitutes the most cost-effective approach, but requires the use of suitably maintained and registered vessels.

Despite the best intentions, it is possible that MINNS will establish before mitigation measures are in place. The notification of Defra, and NRW of any organisms discovered which were not previously extant at the location of the lagoon development is of paramount importance to prevent the spread to other areas.

Through the recommended preliminary MINNS survey, an initial plan for the removal (if required) of any recorded invasive species should be developed, bearing in mind that there is no single method which can be applied to all species. The appropriateness of these plans should be confirmed by NRW prior to being enacted. Early removal of INNS can prevent inundation of a habitat and increase the likelihood that INNS can be prevented from becoming firmly established.

6.7. Monitoring

As part of any intertidal structure construction which may facilitate invasive species, monitoring of biological communities should be conducted pre, during and post project. It is recommended that a series of temporally and spatially replicated surveys (with both fixed and random sampling sites) be carried out at suitable time periods encompassing immediate financial reality against longer monetary outlay to clean up related impact or the loss or impingement of important ecosystem services/related industries. After the pre-assessment, suitable survey periods should be agreed with NRW; nominally every two months as some research suggests that high colonisation levels have been seen over this period (Perkol-Finkel and Benayahu, 2005). Monitoring should occur every three months for the first year following construction and every six months for a further three years. After three years, monitoring should be conducted annually, or bi-annually, for 10-15 years, as artificial reef structures have been shown to demonstrate community shifts after a period of time (Perkol-Finkel and Benayahu, 2005).

6.8. Confidence assessment

In order to assess the confidence in the evidence a confidence table is presented. The confidence table for managing MINNS, and associated biosecurity measures is presented in Table 5.3.14 overleaf.

Table 6.8.1: Confidence assessment of management options for marine invasive non-native species and biosecurity measures.

Keys

Information quality scoring system	1 = Unknown	2 = Inferred	3 = Known	4 = Well known	5 = Well known UK examples

Confidence level	**V. Low 1-2**	**Low: 2.1-3.5**	**Med: 3.6-4.5**	**High: >4.5**

Marine Invasive Non-native Species and Biosecurity		
Factors considered	**Score**	**Notes**
Directly comparable projects or measures	3	There is a range of research that has looked at the colonisation of artificial reefs/man-made structures, (e.g. Langhamer and Wilhelmsson, 2009; Jensen, 2002; Jensen *et al.*, 2002; Boaventura *et al.*, 2006; Hunter and Sayer, 2009). These structures are known to facilitate the colonisation and spread of Marine Invasive Non-native Species (MINNS) (Bulleri *et al.*, 2006; Bulleri and Chapman, 2010). Operation and maintenance can prevent development of a robust diverse indigenous species assemblage and effectively creates an early succession community favouring MINNS (Glasby *et al.*, 2007; Stachowicz *et al.*, 2002). Further MINNS may also be more competitive over a wider range of substrates than native species, and this may result in increased numbers of non-natives on artificial structures (Tyrrell and Byers, 2007; Glasby *et al.*, 2007). The ecology of several foci species are known and mechanisms of spread, and control understood. There are details regarding UK habitats and seawalls. There are numerous ‘best practice’ measures and protocols available and many are currently employed within infrastructure projects.
Level of success	3	Approaches within the literature indicate variable success rates where artificial reef habitats have been colonised and MINNS recorded. Much of the literature details species records and colonisation rates, but the evidence base regarding successful mitigation measures, or eradication programmes, appears low.
Environmental parameters of species or measure understood	4	Details on the potential beneficial features that can be utilised within lagoon walls (use of other rock substrata, construction of rock pools, use and density of bioblocks, complexity of the rock surfaces, etc.) are known and can be used for coastal lagoon developments. Importantly these detail the benefits

		of increasing biodiversity potential to mitigate the colonisation and spread of MINNS. Protocols to manage MINNS associated with construction, operation and maintenance, and de-commissioning are widely available and used on other infrastructure projects. The ecology of foci MINNS is generally well-understood and vectors known, along with control measures, and reporting and control procedures.
Analogous examples	3	Consideration of the colonisation of communities on artificial reefs and other structures within European and other temperate waters can be considered as analogous.
Temperate examples	3	A range of other artificial reefs have been constructed in Europe and in other temperate locations. There are also many thousands of piers or seawalls that have been constructed for harbours or as sea defences, which may provide insight into the recording, control, management, and mitigation for the spread of MINNS.
Tropical examples	N/A	
Total score	**16**	
No. of factors rated	**5**	
Average score	**3.2**	

Confidence level	**Low 3.2**

7. Discussion

7.1. Overview

Tidal lagoon projects are inevitably large and will impose a considerable footprint upon the marine environment. They can be expected to have a range of impacts associated with biotic and abiotic receptors that fall into five broad categories:

- The footprint beneath the entraining wall, sluices and turbine house;
- The footprint of the donor site for the sand core within the entraining walls;
- The lagoon itself, in which patterns of erosion and sedimentation can be expected to change from the preceding conditions;
- Wider coastal process effects caused by disruption of currents and wave patterns and manifested by changes in near-field (and possibly far-field) sedimentation and erosion; and
- Disposal of sediment dredged from within the lagoon and deposited at a licensed disposal site.

Assessment of the impacts is therefore dependent upon a number of critical factors:

- Thorough modelling of the likely coastal process effects, including the application of a conceptual model of the ways in which the structure may be predicted to affect coastal processes;
- Suitable (high) resolution data on the distribution of biotopes and their potential sensitivity to particular pressures such as sedimentation or erosion;
- A clear project plan and a defined design;
- Understanding of the localities of designated wildlife sites and any UK Biodiversity Action Plan (UK BAP) habitats, or those listed as a habitat of principal importance in Section 42 of the Natural Environment and Rural Communities (NERC) Act, 2006;
- Understanding of the likely efficacy of mitigation measures; and
- A clear understanding of what might be proposed as alternative offsetting measures (including specific compensation measures in the event that Natura 2000, Ramsar or SSSI interest is affected).

Rigorous screening and scoping for EIA is essential, and to be effective it must involve early dialogue with competent authorities and statutory bodies such as Natural Resources Wales (NRW). There are several spin-offs from effective EIA, including the development of a Statement of Common Ground between the developer and the Statutory Nature Conservation Organisation (SNCO) and, ideally, key Non-Governmental Organisations (NGOs) such as the RSPB and Wildlife Trusts. This forms the basis for constructive dialogue and the development of a Compensation, Mitigation and Monitoring Agreement (CMMA) between the parties (assuming there is not outright SNCO/NGO opposition to proposals).

A well-constructed CMMA can be expected to include clear analysis of the expected unmitigated impacts (early identification of likely, realistic, environmental impacts is fundamental to the process), what mitigation measures are expected to deliver, and the nature of any need for residual offsetting measures (compensation in the case of Natura 2000 and Ramsar sites). A CMMA should also clearly set out the hypotheses under-pinning any monitoring proposals, and a clear understanding of the approach to be taken in the event that monitoring highlights a departure from the projected hypothesis of the project's performance. Any residual uncertainties would be resolved by employing an adaptive environmental monitoring/management plan (AEMP).

Part of the process leading up to the development of a Statement of Common Ground involves analysis of the sensitivities of relevant marine receptors. This is a challenging area in which the evidence-base is incomplete. MarLIN is often the only available source for assessing the sensitivity of receptors, but it was not developed in the context of risk assessment and enhancement measures. Its application in this arena may therefore not always offer a complete framework for agreement. Nevertheless, it remains the best available science, which may be supplemented by reporting the outcomes of monitoring associated with development projects. The need to develop a better understanding of receptor sensitivity, and outcomes of monitoring programmes, highlights current shortfalls in access to peer-reviewed assessments of biological and geomorphological responses to marine development projects.

It is also important to understand/recognise if/when offsetting, or enhancement, measures are included within a project proposal to facilitate wider community and stakeholder engagement. Where mitigation is not required to fulfil the statutory requirements developers may wish to synergise societal perspectives with those enhancements proposed purely for biodiversity reasons. It will be useful if these additional values are explicitly detailed within future coastal lagoon development projects, if that is the intention. These aspects are relevant, and may have some bearing upon any proposed measures, including enhanced value.

This report is designed to contribute to future engagement between NRW and developers of possible/potential tidal lagoon projects in Wales. It focuses on the levels of confidence that can be attached to measures that might be considered as possible mitigation, compensation and enhancement associated with future tidal lagoon projects.

7.2. Effective measures to offset habitat loss

Development of an effective range of measures to ameliorate the negative environmental impacts of development in the marine environment is still in its infancy. It differs from the terrestrial environment because habitats are often degraded by anthropogenic activities, but are far less likely to have been totally eliminated as has happened through agricultural intensification on land. It is therefore unlikely that biodiversity gains will be realised in the same way as habitat creation is effected on land.

It should also be noted that there are numerous complications associated with the terminology of offsetting biodiversity impacts. The question of compensating for loss in such situations is a largely unexplored concept (with the exception of the Rotterdam Mainport (Maasvlakte II) project in the Netherlands). Mitigation mainly concerns measures incorporated into project design and methodology, but is sometimes used to describe measures to offset impact on undesignated habitat or species. In the case of TLSB a variety of measures are offered as mitigation, some of which appear simply to be a convenient by-product of the construction project. As such, they may, or may not, actually confer any real mitigation (if they achieve what is advanced as a beneficial outcome).

It follows that many of the possible measures that might be incorporated into project design represent 'enhancement' in which a judgment is made that a change in the physical nature of a receptor site represents a biodiversity gain. There is very little guidance to inform what must be regarded as a policy issue, but there are North American examples that might be called upon to inform a developing debate.

In the absence of available examples of UK policy it has not been possible to assess 'enhancement' measures against a rigorous policy framework. This issue remains to be resolved and highlights the urgent need for an informed debate about the types of measures that may be regarded as 'enhancement' and the circumstances in which they might be applied. There are potentially two differing situations:

- Firstly measures within European marine sites (SAC, SCI, SPA and Ramsar sites) that may lead to conversion of one habitat type to another that might be regarded under some circumstances to be more 'desirable'; and
- Secondly, where 'enhancement' measures affect undesignated habitat.

This is a matter that has widespread application, particularly in relation to marine renewable energy projects.

Importantly, there is very little in the literature to suggest that particular marine habitats can be re-created, although there is a strong body of evidence to suggest that restoration measures for some habitats may be possible (e.g. seagrass beds). The most obvious way of addressing loss of biodiversity under the footprint of major marine construction projects is to look at ways in which the biodiversity value of other areas can be enhanced by lifting anthropogenic pressures. In other words, making a judgement as to the condition of the sea bed in other areas, and excluding damaging activities to improve the ‘condition’ of relevant assemblages, such as establishing age-class structure, species numbers and species composition, and community connectivity. Alternatively suitable receptor sites may be identified, to facilitate the translocation of certain discrete communities from impact zones, to an environment able to support and maintain them.

Detailed consideration of secondary effect footprints and pathways associated with the location and lifespan of a lagoon development project (construction, operation and maintenance, and decommissioning phases), are critical. Receptor sites should be located outside the influence of such effects if they are to be viable biodiversity enhancement.

It is important to note that some enhancement methods rely upon the sourcing/harvesting of ‘seed’ or ‘brood’ stock e.g. establishing *Zostera* spp. or *Ostrea edulis* beds. In many cases suitable sources of seeds, rhizomes, spat etc., may themselves be located within existing marine protected areas as qualifying designated features. Use of these features to ‘supply’ enhancement projects may contravene existing conservation objectives, and also constitute permitted or licensed activities in their own right. Further complication may arise where sources of ‘seed’ or ‘brood’ stock are identified from a wider geographical range (to mitigate impacts on local qualifying feature populations) are associated with a different genetic population. Whilst the instances of adverse effects on genetic diversity may be low, this aspect will be considered within a robust application and ES.

A few habitats, such as some saline lagoons and some intertidal assemblages, may be generated by deliberate physical interventions. Outcomes of such habitat creation are far from certain, and can involve several decades of evolution before communities/biotopes become fully representative of a given target. Detailed analysis within this report, using a semi-quantitative assessment process, is presented in a series of confidence tables for key habitat types, whose overall conclusions are presented in Table 7.2.1(reproduced below).

Table 7.2.1: The overall confidence levels for each of the habitats, species and measures assessed in this review.

Habitat/Species/Measure	Confidence score	Confidence level
Saltmarsh	**5.0**	**High**
Intertidal mudflats	**4.2**	**Medium**
Coastal saline lagoons	**5.0**	**High**
Intertidal sheltered muddy gravels	**1.4**	**Very Low**
Seagrass beds	**3.6**	**Medium**
Sabellaria alveolata reefs	**2.5**	**Low**
Ostrea edulis beds	**2.4**	**Low**
Artificial substrata habitat	**3.6**	**Medium**
Non-migratory fish habitat	**2.8**	**Low**
Atlantic Herring spawning habitat	**2.0**	**Very Low**
Marine invasive non-native species and biosecurity	**3.2**	**Low**

This analysis highlights the considerable uncertainties about the likely efficacy of possible offsetting measures. Most confidence can be attached to the likely outcomes of habitat creation involving intertidal habitats behind existing seawalls, or similar infrastructure. In this respect, there is substantial evidence to indicate that some saltmarsh habitats are re-creatable in the long-term, and that coastal saline lagoons can be created that can support a representative assemblage of lagoonal-specialists.

The precise potential for intertidal and saline lagoon habitat creation in Wales is less certain. This is because projects to date have largely been undertaken in England. Such projects have often involved muddy environments rather than the higher energy environments of Wales, where hard substrate or sandy conditions prevail. In the case of projects affecting the Severn Estuary, high suspended sediment levels will have significant implications for the use of managed realignment and saline lagoon construction: saltmarsh will result in both cases without continual intervention. For saline lagoons, active management of man-made systems, via sluices, is proven to mitigate siltation, but 'normalisation' of salinity through regular input of seawater is detrimental for most specialist species and communities. Mudflat creation is highly problematic in almost all circumstances, and there is considerable uncertainty about design measures that might avoid inevitable progression of managed realignment to saltmarsh.

The TLSB project mainly impacts upon hard surfaces in the intertidal zone and upon subtidal habitats where the evidence base for possible offsetting measures is generally poor. It would appear that there is no experience of re-creation of intertidal sheltered muddy gravels. There is also a minimal evidence base to support any contention that loss of Atlantic Herring spawning grounds can be offset by fish using new structures. Any reliance upon the rock armour, or gabions, of the TLSB wall acting as mitigation must be considered with extreme caution given the minimal level of sound evidence extrapolated from analogous situations elsewhere.

Measures to address impacts on Honeycomb Worm *Sabellaria alveolata* reef, Native Oyster *Ostrea edulis* beds, and non-migratory fish also fall into the lower confidence spectrum. The evidence base is weak but it does offer some supporting information from which judgments can be developed.

There is no evidence, beyond Swansea Bay, to support the contention that *S. alveolata* reef can be translocated. Initial trial results from the TLSB pilot study are encouraging, though not proven at the scale required to offset net loss associated with the full-scale of the proposed project impact footprint. It is also noteworthy that in the only analogous example discovered by the project team was a study in which the fan worm *Serpula vermicularis* was collected and replanted in two Scottish sea lochs. In this case, transplanted tubes gradually disappeared and only remnants were left ten months later. The real outcome of the TLSB translocation can therefore only be fully assessed after several years in which recruitment and survival are monitored against a consistent control/reference site. Consequently initial results of the trial translocation should not be considered a reliable indication of the long-term prognosis for translocated *S. alveolata* reef, especially considering the full-scale translocation associated with the proposed project impact footprint. Value from the pilot study will be enhanced if monitoring of the donor reef blocks continues throughout construction, and post-construction (in addition to monitoring proposed for full-scale translocation should TLSB receive consent).

A variety of factors need to be taken into account when considering the possible efficacy of a programme of captive breeding and re-laying of Native Oyster populations. Key considerations include identification of a suitable location for laying new beds within proximity to any proposed coastal lagoon development. Water quality parameters and the need for maintenance dredging within the tidal lagoon mean that the prospect of successful establishment of self-maintaining Native Oyster beds within the lagoon basin is open to considerable doubt. Furthermore, the production of viable spat is a complex process that can be complicated by the source of the broodstock and the ways in which spat is reared. If restoration of Native Oyster is to be considered, it would be wise to minimise the key uncertainties associated with long-term habitat creation within the tidal lagoon itself.

Artificial reefs are a well-established enhancement measure developed in a wide variety of temperate and tropical situations. There are also numerous possible analogues arising from major coastal structures that may help to inform judgment about the possible benefits marine flora and fauna from a structure such as a lagoon with walls with in-built rockpools, inclusion of bioblocks, or formed from rock armour or gabions. The evidence base is reasonably strong considering the opportunities for establishing representation of local marine invertebrate and algal communities, as well as potential to accommodate increased biodiversity compared to the natural baseline. Unfortunately when considering non-migratory fish habitat, published evidence primarily relates to nearshore/offshore structures and installation and not to estuarine fish. Consequently, it is not possible to place any strong reliance on such structures as enhancement for non-migratory estuarine fish. Nevertheless, it remains possible that there may be some positive benefits to certain local fish populations.

Creation of *Zostera marina* seagrass beds is well documented across temperate countries although there is a minimal evidence base associated with the UK. Various seeding and translocation methods have been used, and with relatively high levels of success. Most programmes involving the creation of seagrass beds have an indicated a level of resource cost, with successful measures usually involving the use of SCUBA divers. The most likely constraint to successful seagrass habitat creation or translocation will be the identification of suitable donor and receptor sites. However, this may be the case for most habitat enhancement measures so is not unique to this habitat type. Confidence is recorded as medium for this habitat, but this is primarily a result of the low evidence base for UK-specific examples only. This is especially the case considering the mosaic of *Z. marina* and *Z. noltii* associated with heterogeneous sediment and cobble/boulder habitat within the Severn Estuary.

7.3. Considerations of biosecurity

Disturbance has been shown to result in lower diversity indigenous communities which are less able to buffer change, such as spatially aggressive invading marine species arriving through vessel transport, local area disturbance, or other mediums. It is increasingly evident that legislation to encourage proactive management of non-indigenous invasive species. In the relatively open system of the marine environment this seems to be the most sustainable approach. Reactive, post invasion management is nearly always expensive and may not be a realistic option. The ecosystem service costs, whilst less tangible, are also likely to be considerable.

In assuming that the creation of hard substrate habitat will be beneficial for overall biodiversity, the factors involved will require careful management. Whilst such ‘beneficial habitat’ has in the past been created with statements about biodiversity gain, recent legislative drivers and general awareness reflect changing attitudes towards facilitating the arrival of invasive non-native species. This regulatory environment demands a precautionary approach to managing the risk posed by introducing hard structures, such as lagoon walls, into soft sediment systems.

The use of bioblocks within low diversity estuarine environments will potentially provide little biodiversity gain and may compromise biosecurity. These issues need to be considered on a case-by-case basis and should be discussed with NRW to optimise biodiversity potential without compromising biosecurity. The aim of artificial reef habitat (the lagoon wall) therefore needs to encourage as diverse and natural community as possible. It should be accompanied by frequent monitoring to detect and address problems with invasive species.

7.4. Application to new development proposals

A comparatively exhaustive literature review has demonstrated that uncertainty and lack of relevant case law are the biggest challenges arising from major development projects. Several critical challenges emerge:

- The need for a sound policy basis upon which to make decisions about the loss of undesignated habitat offset by measures that are reliant upon changing the structure and function of existing habitats;
- The need to investigate the potential for habitat restoration at an alternative location to the impacted/lost marine habitat. Such an investigation needs to consider ways of ensuring long-term maintenance of enhanced habitat;
- How to consider 'trials' or 'pilot studies' involving novel methods with low confidence of success? There is a potential for increased scientific knowledge concerning the efficacy of delivery of certain habitat creation schemes, and these should not be discounted outright. However, there is uncertainty associated with the validity of such measures contributing to effective mitigation, offsetting or enhancement. How can these can be 'underwritten' within the process i.e. if they fail, is the developer likely to be penalised?;
- The degree to which risk is addressed in the context of undesignated habitat, including enforceability of delivery of enhancement measures out with designated or notified nature conservation sites, and links to effective AEMP to consider impacts that subsequently fall outside of residual EIA envelopes. This is a policy matter that cannot be resolved by any evidence base, even if one existed; and
- The comparative absence of peer-reviewed analysis of previous development projects. In the absence of readily accessible reports there are limited foundations for developing sound guidance to improve the outcomes of Environmental Impact Assessment, development of enhancement measures or agreement between developers and statutory bodies on the projected outcome of a particular proposal.

It is clear that in the majority of cases, there is an inadequate evidence base upon which to place great confidence in the measures proposed within the TLSB project. Some ideas, such as *S. alveolata* reef translocation, are novel and may prove effective after longer trials and monitoring. Others, such as the incorporation of bioblocks into the structure such as the walls of a lagoon, are dependent upon using sufficient numbers to genuinely modify the biological functionality of the structure. Considerable thought needs to be applied to the level of bioblock extent within a structure that would represent a genuine improvement in its biological performance. This is equally true of the role rock armour and gabions might play in the formation of artificial reef structures that benefit non-migratory fish, and possibly also spawning Atlantic Herring. Current proposals involving the use of granite appear to represent the weakest possible potential, based on existing knowledge of the performance of different rock types as reef structures. The sourcing and use of local rock types can be considered to be a preferable solution.

Considering the expected need for extensive and repeated dredging within the lagoon basin, there is little evidence to suggest that a stable system will be established within such a structure. There will be repeated water quality and biological perturbations, even if undertaken on a rotational basis. Consequently, the potential for generating long-term functional replacement of lost habitat within the lagoon walls is open to considerable doubt. The extent to which this may apply will depend upon local suspended sediment conditions. Consequently coastal lagoon developments built in areas where suspended sediment loads are low will require less maintenance. Conversely, those in high suspended sediment conditions such as the Severn Estuary are likely to require more frequent intervention. These factors mean that it would be unwise to place significant reliance upon habitat creation measures such as laying seagrass or Native Oyster beds within the lagoon as realistic offsets for loss of existing habitat. Such measures in suitable locations outside the development footprint are potentially beneficial and viable under certain circumstances, subject to resolving some of the challenges outlined in the analysis above.

It is important to bear in mind that the distribution of particular biotopes is a function of various physical factors such as exposure to wave action and in particular to storm events, as well as the effects of tidal range, tidal currents, sediments loads and associated turbidity, and salinity gradients. Habitat re-creation within the marine environment is therefore severely constrained by local conditions. It is therefore a matter of developing a clear policy approach to habitat loss and replacement. Such a policy would need to make a judgment of the habitats that are 'least' and 'most' important in terms of possible donor sites. The most obvious and potentially successful 'enhancement' measures are highlighted in Table 7.4.1.

Table 7.4.1: Possible biodiversity enhancement measure in relation to coastal lagoon development projects.

Possible Biodiversity enhancement measures	
Measure	**Current state of knowledge**
Lifting anthropogenic pressures on existing habitats	Currently used in Highly Protected Marine Zones. Implemented in relation to Rottedam Mainport (Maasvlakte II) development.
Re-planting/restoring seagrass beds in locations known to have previously supported such habitat	Well understood in other parts of the World and specifically for *Zostera marina*.
Creation of rocky reef habitat for biodiversity gain and invasive non-native species mitigation	The principles of biodiversity enhancement of artificial reef structures are well known, along with techniques able to support habitat diversification e.g. rockpool creation, use of bioblocks etc. Positive environmental effects from artificial reefs can be achieved through considered planning and for a relative minimal extra cost at the engineering and construction phase. Further it is understood that enhancing artificial reef biodiversity can be an effective tool to mitigate the spread of marine invasive non-native species. Subtidal structures can facilitate positive effects through habitat provision for fish species, although the evidence base concerning estuarine species is scant.
Creation of sluiced coastal saline lagoons in sustainable locations	Several regulated tidal exchange sites around the UK coast. None linked to development projects as yet but agreed compensation for Able Marine Energy Park on the Humber Estuary includes such a measure. Numerous examples of sluiced borrowdykes in Kent and Essex where the dykes have developed a fauna similar to saline lagoons.

Large-scale managed realignment projects that are sufficiently large to develop semi-independent morphological functionality. For example, re-connecting river systems that have been separated from tidal influences.	So far none have been tried, but this approach has been proposed as part of remediation for elevated tidal propagation and suspended sediment levels in the Ems and Elbe Estuaries in Germany.

The analysis presented in this report highlights the need for careful consideration of the circumstances that may evolve throughout the application process, construction, and into operation and maintenance phase. NRW staff will need to be mindful of enhancement proposals set in the context of potential outcomes and actual realised deliverables. For certain habitat enhancement measures it is possible that actual habitat delivery is not entirely what was envisaged. Where this is the case, the reasons for sub-optimal enhancement, or failure to deliver should be assessed and used to inform future projects. It is important to ensure that biodiversity enhancement is not perceived as an easy win that results in very little actual biodiversity gain. Instead, it is desirable to identify effective measures and to ensure that they are delivered, or trialled where appropriate.

7.5. Knowledge gaps

This review has highlighted the general absence of information on marine habitat creation, translocation, and restoration. There are notable exceptions such as the use of no-take zones to restore ecosystems, artificial reef construction in nearshore and offshore locations, managed realignment to yield saltmarsh, and creation of certain types of saline lagoon. Beyond this point, the knowledge gaps fall into five categories:

- Policy examples that explain how the issue of non-like-for-like mitigation and enhancement measures can be incorporated into consents for major development projects in a consistent manner;
- How ‘lifting’ anthropogenic pressures might be brought into the toolbox of measures to offset major development projects;
- The use of anthropogenic structures by non-migratory fish in nearshore and estuarine situations;
- Factors that influence the development of mudflat in managed realignment, and possible designs to ensure mudflat rather than saltmarsh creation; and
- Subtidal sediment habitat impacts/loss is likely to be associated with future proposals for lagoon development projects with attendant mitigation and compensation measures required. Detailed investigation into the potential for creation and/or compensation will be needed.

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Appendix A

Extract from Tidal Lagoon Swansea Bay Environmental Statement Appendix 8.3: Artificial Structures in Coastal Habitats: Optimising the value for biodiversity by creating an artificial reef

To aid Natural Resources Wales staff reviewing tidal lagoon development projects – to be read in support of Section 5.3.7

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Appendix 8.3

Artificial Structures in Coastal Habitats:
Optimising the value for biodiversity by creating an artificial reef

Prifysgol Abertawe
Swansea University

SEACAMS

ERDF
Llywodraeth Cymru
Welsh Government
Ewrop & Chymru: Buddsoddi yn eich dyfodol
Cronfa Datblygu Rhanbarthol Ewrop
Europe & Wales: Investing in your future
European Regional Development Fund

Artificial structures in coastal habitats

Optimising the value for biodiversity by creating an artificial reef

Report for Tidal Lagoon (Swansea Bay) plc

SEACAMS contract	B56
	This review fulfils part of the contract "Determining the key issues related to artificial substrate as a coastal habitat for enhancing marine renewable developments"
Contract Partner	Gill Lock, Eva Bishop, Tessa Blazey **Tidal Lagoon (Swansea Bay) plc** Pillar House 113-115 Bath Road Cheltenham GL53 7LS
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This report was produced by SEACAMS, a project part-funded by the European Regional Development Fund through the Welsh Government.

Summary

I. Artificial coastal structures can supplement the natural environment by providing a substrate that will support marine organisms, thereby functioning as an artificial reef.

II. The structural diversity of the artificial reef will determine the diversity of marine organisms utilising created habitat.

III. Negative effects of artificial reefs can potentially be the provision ‘stepping stones’ for invasive species if little other hard substrate is available. In Swansea Bay this may be less problematic since there is already a considerable amount of natural and artificial hard substrate available. However, monitoring should include checks for invasive species.

IV. Positive effects of artificial reefs include the enhancement of biodiversity of invertebrates and fish. Hard substrates such as rock armour provide attachment surfaces for species that themselves create reefs, for example oysters, mussels or tube-building worms (e.g. the Sabellaria honeycomb worm). These accelerate the reef-building process.

V. Diversity promoting design solutions can be included in the planning and construction phase of the tidal lagoon wall to attract large numbers of species and to reduce the impact on the surrounding environment.

VI. The specific advice for TLSB in this report follows recently produced guidelines which are based on rigorous scientific research.

Recommendations

In order to maximise benefits of the lagoon wall for the environment, the following general principals should be considered:

i. *Avoid smooth rock material.* Few organisms will colonise homogeneous surfaces and species colonisation rates will increase with surface roughness. Where possible, use a mixture of hard and soft rock. Soft rock (e.g. limestone) will erode quicker than hard rock (e.g. granit) which will create surface roughness and habitat for attachment of marine organisms.

ii. *Create rock pools, pits and crevices.* Rock pools, pits and crevices provide refuges for intertidal organisms and can sometimes support greater diversity than emergent substrata. They can be created by drill-coring into boulders.

iii. *Incorporate precast reef units in or around the lagoon wall*. Artificial units that are designed to maximise the number of attracted species promote biodiversity and can provide protection from erosion.

iv. Facilitate 'soft' engineering. They include the creation of saltmarshes, seagrass meadows, oyster and mussel beds and tube-worm reefs (*Sabellaria*). These features offer protection from hydrodynamic impacts, retain sediment and reduce erosion, but they also diversify the habitat.

Summary of recommendations for the TLSB	
1.	The surface of the rock armour should be uneven and as porous as possible.
2.	A mixture of different materials should be used as rock armour. Some of the material could be softer than others, thus allowing different degrees of erosion.
3.	The surface of the rock armour could be modified artificially. Ridges, crevices and depressions could be deliberately chiselled into the lagoon wall.
4.	Artificial rockpools could be added to the rock armour of the lagoon in the inter and subtidal area.
5.	Precast reef blocks such as 'BIOBLOCKs' or 'Reefballs' could be integrated into the lagoon wall to maximise diversity of colonising species.
6.	Gabion-type cages and matrasses filled with local bivalve shells could be integrated into the lagoon wall to attract benthic invertebrate fauna. Shells could include waste from shell-fish processors and restaurants
7.	The slope of the foot of the lagoon wall should be drawn out and shallow.
8.	Habitat properties of the species-rich area around Mumbles Pier should be analysed and as far as feasible re-created at a suitable location outside the tidal lagoon.
9.	A program to create native oyster reefs (*Ostrea edule*) inside the lagoon should be facilitated.
10.	A program that promotes the restoration of honeycomb worm reefs (*Sabellaria alveolata*) should be facilitated.
11.	Experiments to create seagrass habitat inside the lagoon could be supported.
12.	Further opportunities to create multi-species, integrated multi-trophic systems inside the lagoon area should be explored.

1.0 Introduction

1.0.0.1 Tidal Lagoon Swansea Bay plc (TLSB) proposes to construct a tidal energy lagoon. The structure will be made up of a 9.5 km breakwater wall extending out from the Port of Swansea, and it will enclose an 11.5 km² tidal area. The nature of the project requires rocky building materials which will be placed on top of inter and subtidal soft substratum in Swansea Bay. The lagoon wall will add to the existing rocky shore in Swansea Bay. Change in the fauna of the lagoon area is unavoidable since soft and rocky shores are colonised by different species communities, with some overlap. In this report we accept that the presence of the lagoon will alter the nature of the invertebrate and fish community, but it is deemed desirable to optimise the lagoon design so that it functions as an artificial reef. The use of different materials, textures and structures could affect the number of species attracted to the lagoon wall and the biodiversity of marine life it supports.

1.0.0.2 The need for artificial coastal structures is going to continue in the form of sea-defences in response to climate change and rising sea-levels, for coastal developments such as harbours, jetties and pontoons and for marine energy structures, such as oil and gas platforms. Marine renewable energy installations are also becoming much more frequent with the need to shift focus of energy production away from fossil fuels and towards renewable energy. Specifically round 3 and 4 of the Crown Estate offshore wind licence auctions will result in a potentially vast increase in hard substrate around the UK coasts. Recent legislation has put more emphasis on environmental and socio-economic benefits of coastal structures to minimise or mitigate ecological impacts. A guidance report for the Environment Agency as well as partner governmental bodies and developers has been produced with information and advice regarding ecological enhancement in the planning, design and construction stages of hard coastal structures (Naylor *et al.* 2011). Most of the methods outlined are still experimental, although some have been included in constructions as mitigation. The report describes methods of general and specific ecological enhancement. 'General ecological enhancement' includes practises such as arranging rocks in rock groynes to maximise void space for fish and invertebrates to utilise (Li *et al.* 2005). 'Specific ecological enhancement' is used for targeting particular species or habitat niches, such as building rockpools in vertical walls.

1.0.0.3 The perception of desirability or undesirability of effects of artificial structures in the marine environment are value judgements related to societal goals and expectations (Firth et al. 2013). The following management targets were identified (Burcharth et al. 2007): Provision of suitable habitat to promote living resources for exploitation of food (such as shellfish and fish), living resources that are the focus for recreational or educational activities (angling, snorkelling, rock-pooling, bird-watching), conservation of endangered or rare species and rocky substrate assemblages (biodiversity) for conservation or mitigation purposes.

1.0.0.4 The aim of this report is to investigate the ecological value the lagoon wall could provide and to suggest design measures that would benefit the ecosystem. We base recommendation to a large extent on the latest research on eco-engineering of coastal defence structures in the coastal and marine environment (Firth et.al 2012, and Bohn et al. 2013).

2.0 Artificial structures in the marine environment

2.0.0.1 Artificial structures like pontoons, piers, breakwaters or seawalls are increasingly common in coastal waters. While the primary objective is to protect areas from flooding or erosion, or modify hydrodynamics and sediment movement, these structures will be colonised, or 'fouled', by marine algae and fauna. Sessile marine organisms will settle on the firm, stable surfaces, and the artificial structures become new habitats for animals and plants (Ardizzone *et al.* 1989, Baine 2001, Bombace et al. 1994, Chapman & Bulleri 2003).

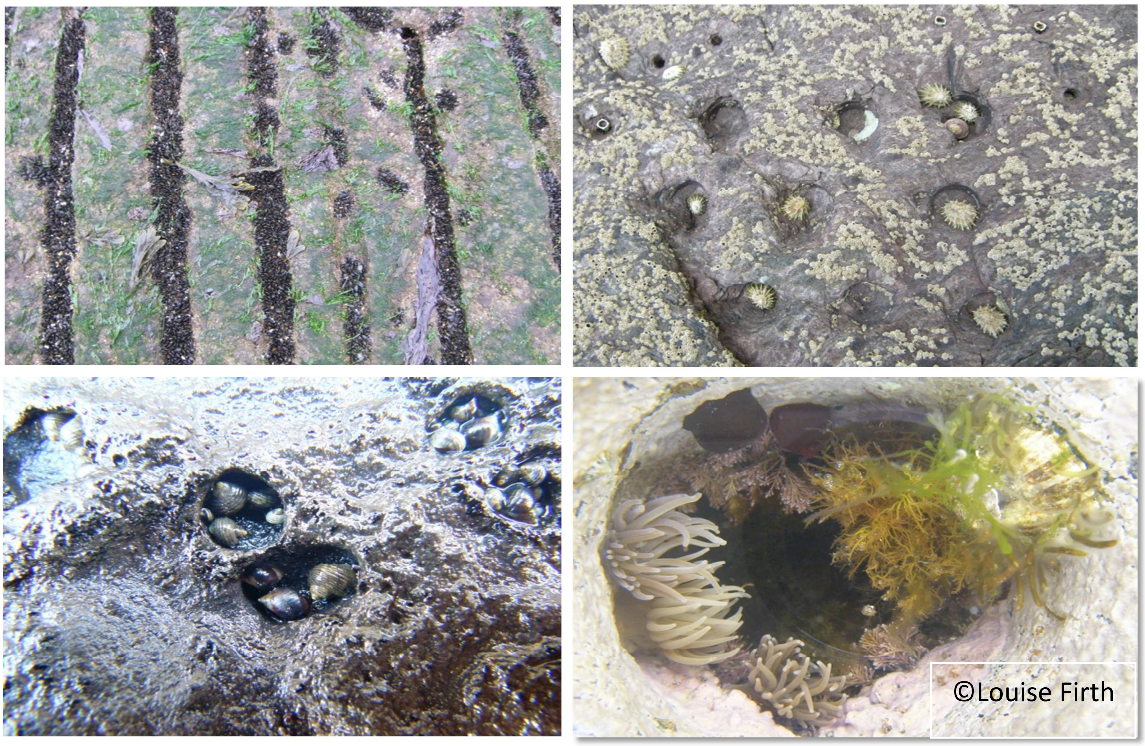

Figure 1: Environmental heterogeneity: Roughness, crevices & pools

2.0.0.2 Depending on the complexity of the man-made structures they have the potential to behave as artificial reefs, providing not only substrate for sessile organisms, but also shelter and protection for motile species such as fish and crustaceans. Generally, the diversity of the biotic community is directly linked to the diversity of the physical habitat, be it natural or artificial (Langhamer *et al.* 2009). Natural weathering and erosion in limestone, for example, can provide crevices and surfaces more suitable for marine organisms to utilise. On a micro-scale, geology and surface roughness have significant effects on the structure and functioning of the colonising assemblage, while on a small to medium scale, crevices, pits and rock pools provide important refuges for many species (Firth et al. 2013; Figure 1).

Figure 2: Top row: *The Vicissitudes* by Jason deCaires Taylor. 26 life-size figures at 5m depth. Grenada, West Indies. www.underwatersculpture.com. Bottom row: *Another place,* Antony Gormley – 100 life-size statues on the beach at Crosby, Liverpool, UK. These intertidal objects have become colonised by the invasive barnacle *Austrominius modestus*.

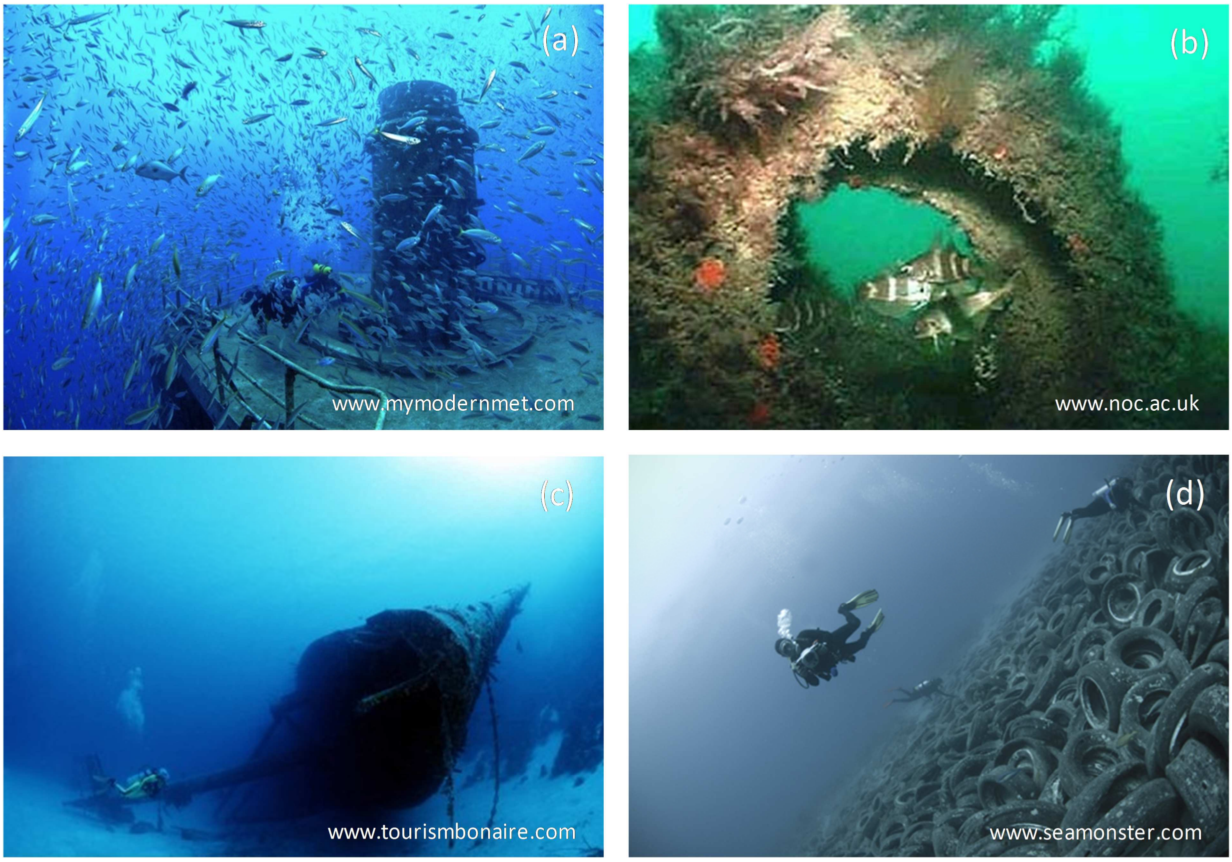

Figure 3 (after Firth 2013). Artificial reefs provided by sunken ships and used rubber car tyres at (a) Key West, Florida, (b) Poole Harbour, UK (c) Bonaire, Dutch Antilles, and (d) Fort Lauderdale, Florida.

2.0.0.3 The nature of the fouling community depends on the specific interactions between the artificial substrate and the local fauna and algae, and the biota of artificial substrata can differ from that associated with natural ones (Connell & Glasby 1999, Pister 2009). Hence, artificial habitats do not substitute natural substrate, but they potentially supplement the natural habitat by supporting naturally occurring marine species. Marine species do not differentiate between the purposes of artificial structures, which was exploited in several arts projects (Figures 2&3).

2.0.0.4 The proposed tidal lagoon walls are to be made from geotubes filled with dredged sand and from rock armour. The latter will provide hard substrate that will be colonised by algae and sessile invertebrates. The lagoon has the capacity to provide a habitat similar to natural rocky shore substrate (Naylor *et al.* 2011). Hard structures like rock rubble breakwaters are considered to have the greatest potential for ecological enhancement for species characteristic of intertidal rocky shores (Naylor *et al.* 2011).

3.0 Negative effects of artificial structures

3.0.0.1 Artificial structures are often built in sedimentary environments that lack rocky shores or other hard substrate. Assisted by the structure, species may colonise an area and get a foothold which would naturally not be the case. The artificial structures will provide stepping stones for the expansion and distribution of species. This may not be desirable. Warm-water species, for example, reached natural shores of the Isle of White by using artificial coastal defence structures as intermediate steps (Keith et al. 2011, Mieszkowska et al. 2006). Artificial structures also appear to be more susceptible to invasive species than natural habitats (e.g. Buschbaum et al. 2012). Individual species may be favoured with knock-on effects on communities which could differ from natural assemblages, and they may even influence the biodiversity of surrounding areas (Inger *et al.* 2009).

3.0.0.2 The proposed tidal lagoon would be situated in an area rich in natural and artificial hard substrate. While the lagoon wall would extend the provision of hard substrate, it seems unlikely that it would provide a new stepping stone for sessile organisms. However, some studies have found that artificial substrata had higher abundances of non-native species. Examples include the green alga *Codium* fragile in the Adriatic and the colonial tunicates *Botrilloides violaceaus* and *Botryllus scholosseri* in Maine, USA (Tyrrell and Byers 2007) in comparison to adjacent natural substrates. Around Swansea Bay, non-native species that feature in relatively high numbers include the slipper limpet *Crepidula fornicata* which competes with the native oysters *Ostrea edulis.* It is also feared that the Pacific oyster *Crassostrea gigas* may invade the Welsh coast and cause changes in the ecosystem similar to other European coasts (Diederich 2005). The lagoon structure would provide attachment material for this alien species, similar to natural hard substrate. Currently there is just anecdotal evidence of individual records of the Pacific oysters, and no significant numbers have so far been detected. We cannot foresee an increased risk of the Pacific oyster colonising Swansea Bay due the lagoon development, other than the additional provision of hard substrate.

3.0.0.3 It should however be considered that the existing rocky shores and artificial structures are already colonised, and that ecological processes such as competition for space and food by fouling organisms have taken place. The new surfaces will provide an opportunity for species to expand their population, and this process should be monitored carefully with the awareness that under favourable circumstances invasive species may take the opportunity to get a foothold.

4.0 Positive ecological effects of artificial reefs

4.0.0.1 The ecology of artificial coastal structures is often compared to that of natural rocky shores, being the closest equivalent in nature (Naylor *et al.* 2011), although communities on artificial hard substrate may be less diverse compared with adjacent natural habitats (Pister 2009). Importantly, artificial structures are often colonised by species that create their own habitat, such as kelp, mussels or honeycomb worms. As a consequence the artificial structure becomes host to secondary biogenic, reef-forming species which accelerate the development of a diverse algal and faunal community.

4.0.0.2 The intertidal zone is a challenging environment for marine organisms to live in. Hydrodynamic forces, emersion and fluctuations in salinity and temperature require rocky shore species to be well adapted to survive in harsh environmental conditions. Around the Bristol Channel the tidal range is high and intertidal animals and plants tend to exist in spatially discrete zones, determined by biotic and abiotic factors. The zonation pattern on the shore is predominantly determined by the tolerance of organisms to being out of water and exposed to terrestrial conditions. The higher up the shore, the longer the period of emersion and the greater the risk of desiccation. The inclusion of crevices overhangs and rockpools in the tidal lagoon wall will increase water retention, and such structures will enable some lower shore species to exist higher up the vertical gradient of the eulittoral zone. The subtidal part of the structure which is always covered by the sea can enhance commercially important species such as lobsters and crabs. These benefits could also be extended to other commercial species such as mussels (*Mytilus edulis*), and native oysters (*Ostrea edulis*).

4.1 Fish

4.1.0.1 Artificial structures in marine environments have the potential to increase diversity and abundance of fish and crustaceans. This has been shown for offshore structures ranging from wind turbines, wave power devices to oil jetties (Langham & Wilhelmsson 2009, Langhamer 2012). Positive effects can be found for fish such as cod, but particularly also for edible crab (*Cancer pagurus*) (Langham & Wilhelmsson 2009). The increase in fish and crabs may have negative knock-on effects for other fauna such as starfish, which are exposed to higher predation pressure (Langham & Wilhelmsson 2009). Positive effects on fish may not just be found comparing artificial reefs with soft bottom habitats, but artificial reefs may also attract a more diverse fish fauna than natural ones (Rilvo & Benayahu 2000). Benefits for fish may result from reefs being enhanced feeding grounds as well as shelter from predators, water movement and trawling. Rilvo & Benayahu (2000) suggested that artificial reefs also provide more successful recruitment grounds. In the longer term a spill-over of fish and invertebrates to other areas further afield would be of benefit to commercial fisheries (Langhamer 2012).

4.1.0.2 There are, however, also examples where artificial reefs had no effect on fish (McGlennon & Branden, 1994), and it is uncertain whether the increase in fish is 'attraction' or 'production'. Ongoing discussions question whether artificial reefs attract and concentrate existing individuals, with no overall net increase in abundance, or whether they actually promote recruitment and the production of individuals, resulting in a net increase of fish or invertebrates (Brickhill 2005, Pickering & Whitmarsh 1997).

4.1.0.3 The variety of artificial reefs that have been studied is vast, making it difficult to assess and compare design aspects and performance as suitable habitats for fish and crustaceans. The difference in geographic location and localised environmental conditions will also have an effect on the species attracted (Baine, 2001). Low-crested coastal defence structures (LCS) may be the most appropriate comparison structure to the Tidal Lagoon Swansea Bay in terms of its construction design. Comparisons of LCS around Europe including the UK, Italy and Spain have identified the effects these structures have on sediments and mobile fauna. These structures have not been found to increase overall diversity in the area, but create a substrate for the development of local assemblages. LCS built in coastal areas dominated by soft substrata can have a strong effect in the structure of the fish community by attracting species typical of rocky shores, and thereby locally increasing diversity (Martin et al., 2005). The fish observed on the structures mainly consisted of juveniles and individuals no older than 2 years, so they do not perhaps offer appropriate habitats for adult fish populations. The accumulation of drifting algae around LCS was shown to indirectly enhance the settlement of fish and crustaceans. This algal detritus can be attractive to new settlers and juveniles of fish and crabs (Martin et al., 2005). Beside the natural accumulation of drifting algae that could occur around the outer wall of the tidal lagoon, small adaptations that mimic drifting algae could have a positive effect on the numbers of juvenile fish recruited to the area. One study found that artificial reef blocks with unravelled polypropylene rope streamers attached attracted significantly higher numbers of juvenile fish (Gorham & Alevizon, 1989). There is the possibility of achieving this with little cost in the way of materials and the labour involved to unravel the ropes and attach them around the structure. Biodegradable materials could be used instead of polypropylene; ropes made from coconut fibre (coir) could be a feasible option. Over time the walls will probably be colonised by macrophytes and will naturally produce drifting algae.

4.1.0.4 Large artificial reef units with varying complexity have been found to attract different compositions of fish species. In a study off the coast of France, reef units that had been filled with materials to increase complexity were found to attract different species of fish (Charbonnel, 2002). Fewer planktivorous fish but significantly higher numbers of commercial species were found in the more complex reef unit structures. The principals described for invertebrates are transferable to fish: an artificial structure that aims to increase fish biodiversity needs to be as heterogeneous as possible. Using a complex mixture of materials such as hollow bricks, concrete pipes etc., creates irregular and interconnected spaces that can be utilized by predatory and prey fish (Charbonnel, 2002). If the tidal lagoon aims to design part of its structure to become conducive to fish and crustaceans, the ecological requirements of the local species should be taken into account. More complex and heterogeneous structures will provide more shelter for fish and crustaceans , with high-profile structures attracting pelagic fish and low-profile, bottom reefs with extensive void space will attract mobile shellfish (Baine, 2001).

4.1.0.5 Langhamer (2012) pointed out that there is still significant research effort required to predetermine specifically the eventual advantages when creating new habitats and how they can affect commercially interesting species such as fish, lobsters and crabs. Aspects of the design of a structure to perform as an artificial reef will need to be influenced by the ecology and the behaviour of the target species (Polovina & Sakaie, 1989).

5.0 Design features of the lagoon wall: creating an artificial reef

5.0.0.1 The design of artificial coastal structures has a major effect on the natural environment (Firth et al. 2013). The magnitude of the effect appears to be heavily dependent on the nature of the created structure, the location and the composition of the native flora and fauna at the time the artificial structure is created. The structural complexity of the building materials and the architecture play an important role for the number and diversity of animals and plants colonising the artificial material, and hence whether an artificial structure will eventually qualify as a reef. Here we review design features that could be implemented in the construction of the tidal lagoon Swansea Bay. The role of the surface texture of building materials is explored, as well as the creation of artificial rockpools and the introduction of building blocks specifically designed to attract a range of colonising species.

Figure 4: Habitat enhancement: drill-cored rock pools at Tywyn. (Firth et al. in press, Coastal Engineering, Photos L. Firth)

5.1 Surface texture: ridges, overhangs and rockpools

5.1.0.1 By mimicking a rocky shore with a mixture of rock sizes, roughness and crevice sizes, then marine life can be encouraged to develop (Li *et al.* 2005). Incorporating porous, calcite rich materials can provide habitat for other organisms, especially rock boring species. This can improve the habitat by increasing the roughness of the materials via bioerosion, which will then be exploited by other species. Also, materials that have a variety of vertical and horizontal surfaces that retain water at low tide will encourage intertidal species to colonise (Naylor *et al.* 2011). Studies have shown that the gradient of substrate can furthermore affect species composition. Most naturally occurring rocky shores have a more gentle slope in comparison to artificial structures like sea walls. Studies have found that vertical substrates support fewer mobile marine organisms (Glasby 2000; Chapman & Bulleri 2003). Surface characteristics such as texture, complexity, size and even colour have been found to affect the numbers and types of organisms that colonise artificial substrates (Glasby 2000).

5.1.0.2 An extension of the modification of surface textures is the creation of artificial rockpools. These can be either attached to the artificial structure or they are excavated from the rock armour. Chapman and Blockley (2009) added experimental rock pools to a seawall, which attracted sessile and motile species, particularly in upper intertidal areas (Figure 4&5).

5.1.1 Outreach: Integrating science, artists and sponsorship

5.1.1.1 The artificial diversification of the lagoon wall would provide an opportunity to actively involve several groups of society and to integrate many interests. The shape and size of pits, ridges and crevices should primarily be diverse, and this could be exploited creatively. Artists could be given the opportunity to design modifications of the rock armour or to sculpture features that could be integrated into the wall. They could collaborate with ecologists in order to take colonisation patterns into account. Sponsors could apply for specific features to be carved into the rock armour. The modifications would also be an opportunity for scientist to test specific features such as standardised rockpools in a controlled manner.

Figure 5: Habitat enhancement pioneers: Artificial rock pools on Sydney seawalls

Figure 6. Pre-fabricated reef blocks; a) Reefballs are used to promote biodiversity (www.reefball.org); b) BIOBLOCK (Firth et al. 2012)

5.2 Precast reef blocks

5.2.0.1 A method to improve the structural complexity of an artificial reef is the deployment of precast units. These are designed to attract particular species or to offer multiple habitat types. A recent example is a collaborative project between SEACAMS Bangor, Conwy Council and Ruthin Precast Concrete (RPC). Bangor University have developed a large-scale habitat enhancement unit called the BIOBLOCK. The objective of the BIOBLOCK is to provide additional habitat types that can be incorporated into rock armour, breakwaters groines and revetments at the construction stage. The BIOBLOCK has rockpools, circular pits and longitudinal depressions.

5.2.0.2 "Reefballs" are based on a similar idea but are designed differently (Figure 6). Depending on the artificial environment and the enhancement unit in question, these can be deployed either during construction or retrospectively to effectively increase local biodiversity (Chapman & Blockley 2009, Firth et al 2013).

5.2.1 Outreach: design competition for precast reef blocks

5.2.1.1 TLSB is considering incorporating pre-fabricated reef-type modules into the lagoon wall and would be looking for the design that is particularly successful at attracting species in Swansea Bay. It is suggested that a public competition could generate a new design of a pre-cast module, or a number of different solutions may result from the competition. Various groups could be invited to the competition, for example engineering students, biology students, schools, investors, general public and artists. The event would strengthen stakeholder engagement and sense of identity with the lagoon. The most successful design would be produced and used in the lagoon wall.

5.3 "Shell BioReefs"- shell filled gabions

5.3.0.1 In recent years engineering solutions have been explored that prevent erosion of coastal habitats, and in particular the loss of sand and mud. In the Netherlands gabion cages filled with fished-up oyster shells were mounted on silt in intertidal areas. A continuous artificial oyster shell reef was created, 200 metres long and ten metres wide (Figure 7). Studies at Swansea University in collaboration the bioengineering company Salix showed that shell filled mesh bags attract a diverse coastal fauna within a short period of time, and they have the potential of enhancing local biodiversity. Shells could include cockle and mussel shells from local shellfish processors and restaurants, as well as dredged shell material from Swansea Bay from within the footprint of the tidal lagoon. The research is currently up-scaled to improve the construction of shell gabions and mattresses, improve their environmental sustainability, explore their effect on the surrounding environment and understand the longer term benefits for biodiversity. These shell bioreefs can potentially be tailored to specific needs of an artificial structure to maximise the environmental benefits.

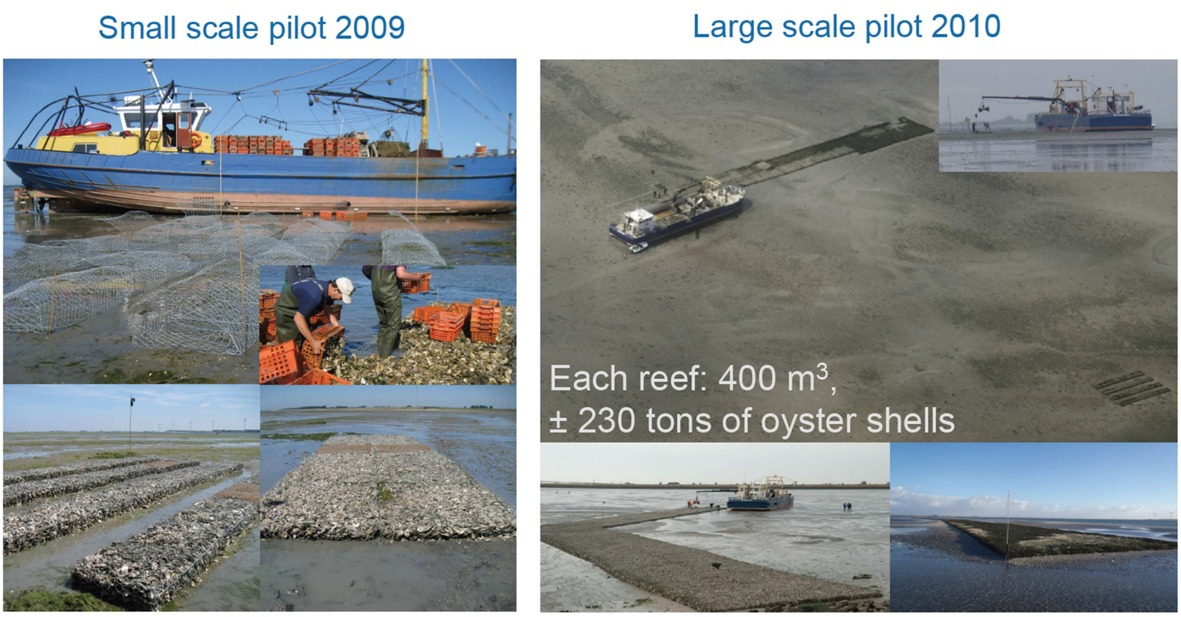

Figure 7: Shell filled gabions in intertidal areas to prevent erosion.

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