What are the benefits of assisted versus natural recovery?

Development of a decision making framework

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Executive summary

Natural England, supported by the Government's 25 Year Environmental Plan, are committed to supporting the delivery of net gain. Currently, under the Environment Bill, biodiversity net gain does not include projects situated below mean low water. However, there is the potential to extend biodiversity net gain to include all development within the England's territorial waters. An important aspect of habitat management (as well as potential to achieve net gain) is understanding and potentially managing the recovery process. This project has reviewed assisted recovery options and options to achieve effective recovery following cessation of impacts or to achieve recovery objectives. The project outputs consist of this report and a standalone Excel spreadsheet which presents an overview of the work and provides part of an overall framework to support decision making around assisted recovery options.

Objective 1 Define recovery

The first objective of the project was to review and define recovery. This project uses the terms natural or passive recovery to describe the potential of a habitat and/or species assemblage to move towards a recovered state following the removal of pressures. The inherent ability of impacted communities to recover following the removal of pressures is referred to as recovery potential. Active or assisted recovery, refers to the application of interventions or measures to initiate or maintain recovery towards but not necessarily to complete the transition to a recovered state.

Objective 2 Assess natural recovery potential

Secondly, the project determined the recovery potential of subtidal habitats (classified according to the EUNIS habitat classification), the project used recovery scores from the Marine Life Information Network (MarLIN) project and information from other sources. For those habitats that have longer recovery times than two years (where recovery is assessed as Medium, Low or Very Low) a literature review of assisted recovery options available was conducted (Objective 3).

Objective 3 Assisted recovery options

Active or assisted recovery options for the marine environment can be classified as either eco-engineering options that improve physico-chemical factors and processes (including sediments, water quality and quantity) or those that engineer the ecology, by replanting or restocking species. Elements of both approaches may be required to assist recovery. Assisted recovery approaches focussed on ecology have been undertaken for a limited number of habitats and species and have typically focussed on biogenic, habitat forming species which play key roles as eco engineers in modifying the environment and that provide additional ecosystem services and goods and benefits.

Objective 4: Assisted recovery costs, benefits, risks, challenges and uncertainties

For each assisted recovery option identified, we provide an overview of the advantages and disadvantages of each when compared to natural recovery, and costs summarised in the technical appendices.

Most approaches to assist recovery of marine species can still be regarded as under development in terms of application to English subtidal marine habitats. Application of most approaches is largely experimental and small-scale although seagrass and oyster restoration projects are beginning to be implemented in larger areas.

A key aspect of feasibility is identifying sites where assisted recovery may be successful, particularly when creating biogenic habitats that may have been absent for a long-time. Feasibility investigations require a range of assessments, including environmental conditions and habitat suitability evaluations. Where transplantation is used, donor populations should be matched as closely as possible to conditions in the transplant site to support population establishment and resilience.

Assisted recovery projects are typically complex, resource intensive and costly. Assessing the costs of marine restoration consistently across methods and habitats is challenging as many studies do not provide complete information on costs and cost reporting is inconsistent. To allow basic cost comparisons, costs sourced from the evidence review were standardised to hectares. Many of the costs reported are for small-scale experimental studies and it is not clear how these may scale-up across larger scale restoration activities where costs per unit area may be lowered. Nevertheless, per hectare costs are consistently high for approaches that involve translocation or reseeding of biogenic habitat forming species. Costs are lower for sediment focussed approaches involving dredging, capping, gravel and shell-seeding that require one-off operations.

A key challenge for assisted recovery approaches is the establishment of eco-engineering species. Where these are present in usual densities as beds or reefs they provide positive environmental feedbacks or settlement cues that encourage recruitment and maintain populations. Biogenic habitat forming species such as kelp, seagrass and bivalves, stabilize and trap sediments and dampen wave energy facilitating retention of larvae and juveniles. Bivalve larvae are typically induced to settle by the presence of individuals of the same species. Where populations are lost, translocated individuals are typically either too small, too sparse or too unstable to modify their environment and establish such self-facilitating feedbacks.

A number of risks are associated with assisted recovery. Approaches may impact donor populations (where stock or transplants are obtained, for example seagrass) and impact habitats and species within the footprint of the recovery project. Biosecurity risks around introduction or spread of pathogens and invasive non-native species are important for a range of projects that involve the movement of stock (transplantation and translocation) or infrastructure such as artificial beds. The creation of permanent or temporary infrastructure at sea will affect activities and other users and may impact safety.

As knowledge and experience of overcoming limiting factors and experience increases, assisted recovery feasibility is likely to improve. Cost-effective approaches that can be used over wider areas are being trialled, such as green gravel for kelps and seed bags for seagrass.

Objective 5. Decision support framework

Many of the decisions around restoration projects and the specific actions to be undertaken are site-specific and cannot be accounted for in a generic framework. Nevertheless, we developed a five step framework to support advice and management options for Natural England that could support decision making workflows. Step 1, to determine the type and source of impacts is site-specific and could not be addressed within a generic framework approach. Habitats within the UK Marine Habitat Classification that occur in inshore and offshore regions of England were assessed against: step 2: assess natural recovery potential; step 3: identify relevant assisted recovery options, and step 4 evaluate feasibility, costs and benefits.

Summary

Marine assisted recovery options focussed on species are expensive and labour intensive. For most marine species, these barriers, coupled with low levels of economic return mean that no options have been developed to assist recovery. It is therefore likely that for most habitats removal of pressures to support the recovery of degraded habitats and management and conservation of remaining habitats will be prioritised over assisted recovery.

Given the low feasibility, high costs and resources required for available assisted recovery options, it is likely these would only be considered where pressures have been removed but populations of high-value species are unlikely to recover naturally due to loss of connectivity and changes in habitat conditions (for example, negative feedbacks).

Assisted recovery approaches have clear value for restoring biogenic habitats that have undergone historic declines and which have not recovered naturally and that on their recovered state provide high levels of ecosystem services and goods and benefits. Significant barriers remain to assisting recovery including costs, feasibility, the complexity of projects and the level of resources required. However, approaches are being developed such as the hessian bag planting method for seagrass seeds and the green gravel approach for kelps, that are lower in cost and scalable to larger areas.

Contents

Introduction

Natural England, supported by the Government's 25 Year Environmental Plan, are committed to supporting the delivery of net gain. Currently, under the Environment Bill, biodiversity net gain does not include projects situated below mean low water. However, there is the potential to extend biodiversity net gain to include all development within the England's territorial waters. An important aspect of habitat management (as well as potential to achieve net gain) is understanding and potentially managing the recovery process. A greater understanding of how to achieve effective recovery following cessation of impacts or to achieve recovery objectives within Marine Protected Areas (MPAs) is, therefore, required and forms the main objective of this specification.

Background

Subtidal marine habitats within UK waters are increasingly affected by several stressors including, but not limited to fishing, aggregate dredging, oil and gas extraction and the expanding offshore wind sector (driven by UK Net Zero ambitions). Consequently, there is a need to inform the sustainable management and conservation of subtidal marine habitats. A critical aspect of habitat management is managing the recovery process; this applies not only following specific stressors such as the construction, operation and decommissioning of Offshore Wind Farms or aggregate extraction sites, for example, but is also required to achieve recovery objectives within MPAs. In order to maximise recovery potential and minimise recovery timescales, Natural England wish to develop a consistent approach to the advice that they give regarding mitigation and recovery options for individual projects, as well as a standard approach to managing subtidal species and habitats with recovery objectives within MPAs.

Coastal and marine ecosystems restoration to date has focussed on biogenic habitats: seagrass beds, coral reefs, salt marshes, oyster reefs, and mangrove forests. The ideal aim of many ecological restoration projects is to return the system to its past natural state. Alternatively, the goal of restoration may be to bring the target habitat to a healthier state (i.e., a "self-maintaining, vigorous, resilient state to externally imposed pressures, and able to sustain services to humans…"; Tett and others, 2013). Under other circumstances, restoration may focus on repairing the structure and function of degraded systems to some extent (see Dobson and others 1997 and Elliott and others 2007 for different definitions) or providing some function where missing (for example, ports or other marine urban environments; Dafforn and others 2015).

Project Objectives

The project comprises the following objectives.

Objective 1. Owing to the high degree of variability in the use and interpretation of the term 'recovery', a definition of recovery in the context of both development and MPA management is required to underpin the subsequent objectives.

Objective 2. Provision of an assessment of subtidal habitat and species recovery potential, with a particular (but not exclusive) focus on the following habitats subtidal sands and gravels (European Nature Information System (EUNIS) A5.1 and A5.2, including Habitats Regulations feature H1110: subtidal sandbanks), subtidal mud (EUNIS A5.3 including A5.361 and A5.362), subtidal mixed sediments (EUNIS A5.4), sublittoral biogenic reefs (A5.6 including Habitats Regulations feature H1170) and subtidal rock (A3 and A4).

Objective 3. For those habitats with anything other than 'High' recovery potential (according to the Marine Life Information Network (MarLIN) definition), a literature review of assisted recovery options available should be conducted (including any recent technologies and methods that have been developed), and where possible, examples of the effectiveness of artificial assistance vs natural recovery provided for each broadscale habitat or feature.

Objective 4. For each assisted recovery option identified, provide an overview of the advantages and disadvantages of each when compared to natural recovery, and include a cost and benefit analysis. These should include but not be limited to time, genetic diversity, potential introduction of invasive and non-natives, and cost of introduction. Assess the feasibility of assisted restoration of all marine and coastal habitats and species considered and summarise the risks, challenges and uncertainties. Provide recommendations for future research/work required where relevant.

Objective 5. Provide a concluding framework to facilitate a consistent standardised approach to the advice and management options considered by Natural England going forward.

Report structure

The report consists of this introductory section and separate chapters for each objective. The report finishes with discussion and conclusions. A glossary is provided and appendices three to eleven contain supplementary information on specific recovery approaches.

A separate Excel spreadsheet is supplied that provides the Objective 2 recovery scores and information and the Objective 5 decision making framework.

Objective 1 Defining recovery

The literature on recovery in marine and estuarine environments is substantial. This project provides a review and definition of key concepts used to refer to recovery and restoration approaches and a glossary of relevant terms.

As outlined in Mazik and others (2015), recovery has been described as the process of returning to a normal state or recovery end point after some period of being degraded (Borja and others, 2012; Tett and others, 2013). Recovery is therefore the process of moving to a state that is considered to be 'recovered' (Mazik and others, 2015). The changing trajectory of a community is the process of 'recovery' and the long-term stability of the climax community at a recognised pristine, reference or target condition would be considered a 'recovered' state (Mazik and others 2015). Recovered status is generally considered to have been achieved when a set of defined recovery end points have been achieved. This definition of recovery is applicable to habitats, species and communities.

The Marine Life Information Network (MarLIN) project, defines recoverability as a potential: "the ability of a habitat, community or individual (or individual colony) of species to redress damage sustained as a result of an external factor". Conceptually, two factors important to recoverability are resistance and resilience, although as outlined by Elliott and others (2007), the term resilience may be used in the same way as resistance by some authors (a key example is Holling, 1986 but see also Peterson, 2000). Resistance, as used by the MarLIN project and other studies to assess habitat and species sensitivity, is defined as the degree to which a variable is changed following perturbation (Pimm, 1984) and describes the tendency to withstand being perturbed from the equilibrium (Connell and Sousa, 1983). Whereas resilience is defined as the ability of an ecosystem to return to its original state after being disturbed. A habitat that has high resistance to a pressure will change less and would generally be assumed to recover more rapidly or more readily from pressures than a habitat which has low resistance and is more changed following exposure. Resistance as a property of ecosystems may vary between pressures. A habitat and biological community, for example, may be very sensitive to abrasion but not to changes in temperature. Recovery may also vary depending on the pressure type and the components of the habitat affected. For example, hydroids associated with soft rock would recover more quickly from abrasion than the substratum would from physical damage.

It is challenging to determine when change has taken place, what status represents recovery, and at what stage of that trajectory a habitat or species population may be. Baselines are dynamic and change through time and may not be reached simultaneously for different indicators of ecosystem status (Borja and others 2010). Borja and others (2012) reviewed the main methods for establishing baselines (or end points) that represent the recovered state. These end points can be, according to their value as a methodology for establishing baselines:

1. pristine conditions;
2. historical observations;
3. modelled predictions; and
4. best professional judgement.

The indicators selected for assessing recovery of species, communities and habitats have included assessments of ecosystem structure, ecosystem function, ecosystem services and other standards (Duarte and others, 2015, Baggett and others, 2015). Mazik and others (2015) suggest that:

"whether defined according to targets, reference conditions or historical baselines, 'recovered' should refer to stability and long term sustainability, within the constraints of natural habitat evolution and variability. That is, species richness, abundance and biomass values (total abundance for a community or an individual species) should be restored together with the component species, their relative abundance and their population structure and that this 'recovered' species or community should be stable (again, within the constraints of natural variability) and sustainable (the biological component has sufficient size and resources to maintain itself over time)."

As Mazik and others (2015) outline, practical workable assessments of recovery and recovered for habitats and species will need to be based, in part, on the best available information and will need to allow for consistent, economically viable and fit-for-purpose monitoring over a realistic timescale. The state of a feature that can be considered to represent a recovered end point should be considered as a range of expected values for a particular feature, derived from historical data (where available) and/or according to existing data from a broad range of sites supporting that feature. A recovering population or community would then show signs of developing towards this range whilst a recovered population or community would remain within this range, subject to natural variability. The required elements to develop a Recovered Reference Range are:

1. long term stability (within the constraints of natural variability);
2. achievability in that unrealistic targets should not be set in terms of, for example, timescale, spatial extent or density given that, for some species and habitats, reaching 'recovered' status may not be achievable;
3. spatially explicit;
4. measurable/quantifiable;
5. contain enough descriptive data to capture the various ecological dimensions of the feature ensuring that the attributes necessary for stability are accounted for (for example population structure as well as species richness and abundance; reef integrity as well as spatial extent);
6. representative of the required legislative end point (for example, favourable conservation status);
7. self-sustaining (i.e. self-sustaining based on interactions between connected subpopulations of a meta population);
8. based on the best available information.

Mazik and others (2015) advanced the definitions of recovery and recovered for species with regard to reference ranges determined on the criteria above, as provided in Table 1.

Table 1. Definitions of recovery/recovering and recovered for species, communities and physical habitats from Mazik and others (2015).

Component	Recovery/recovering	Recovered
Single species	A consistent trajectory, detectable above systemic variability, of net population growth, with biomass and structural population parameters, towards a range of values, specified by the Recovered Reference Range, for a defined spatial area.	A stable, enduring similarity, detectable above systemic variability, of population size, biomass and structural population parameters to the range of values, specified by the Recovered Reference Range, for a defined spatial area.
Communities (multiple-species)	A consistent trajectory, detectable above systemic variability, of community descriptive parameters towards a range of values, specified by the Recovery Reference Range, for a defined spatial area.	A stable, enduring similarity, detectable above systemic variability, of community descriptive parameters to the range of values, specified by the Recovery Reference Range, for a defined spatial area.
Physical habitats	A consistent trajectory, detectable above systemic variability, of a representative set of physical and chemical habitat parameters towards a range of values, specified by the Recovery Reference Range, for a defined spatial area.	A stable, enduring similarity, detectable above systemic variability, of a representative set of physical and chemical habitat parameters to the range of values, specified by the Recovery Reference Range, for a defined spatial area.

Recovery towards a reference range may take place through natural or passive recovery, where alleviation of pressures that have led to reduction in condition (degradation) are removed to allow the habitat and characteristic species to recover (through migration, recruitment) (see Objective 2). Alternatively, active recovery may be desirable or required. Active interventions take a number of forms as described for Objectives 3 and 4. Allowing recovery by removing pressures, taking limited measures to alleviate impacts or enhance biodiversity, or actively restoring ecosystems can be viewed as a continuum or as intersecting approaches that are second-best options compared with conserving high quality near pristine habitats with their natural biodiversity and ecosystem processes (Geist and Hawkins, 2016). Elliott and others (2016) differentiated between restoration based on eco-engineering that improves the physico-chemical processes (including water quality and quantity), and approaches that engineer the ecology, by replanting or restocking species.

This project uses the terms natural or passive recovery to describe the potential of a habitat and/or species assemblage to move towards a recovered state following the removal of pressures, although it is recognised that full recovery may not be possible. The inherent ability of impacted communities to recover following the removal of pressures is referred to as recovery potential. However, it recognised that for marine habitats, determining an unimpacted state against which to measure recovery trajectory and the recovered end point (or Recovered Reference Range) may be limited by evidence. Often for marine habitats there is the issue of shifting baselines, altered stable states or regime shifts and variation between locations in habitat conditions (for example temperature, or the underlying geology) and the structure and function of species assemblages.

Active or assisted recovery, refers to the application of interventions or measures to initiate or maintain recovery towards but not necessarily to complete the transition to a recovered state. This definition is comparable to the term ecological restoration, defined as an “intentional activity that initiates or accelerates the recovery of an ecosystem with respect to its health, integrity and sustainability” (Society for Ecological Restoration). Approaches that mitigate impacts may support future recovery but are not themselves approaches to assist active recovery.

There is some overlap between these approaches. For example, actions (assisted recovery) to restore the habitat conditions would then support the passive recovery of the associated biological assemblage via larval dispersal and migration of mobile species.

Objective 2 Subtidal habitat and species recovery potential

As outlined in Objective 1, recovery represents the trajectory towards a recovered end point. The main factors influencing recovery of features were identified by Mazik and others (2015) as: (i) initial and ongoing pressure extent, intensity and frequency, (ii) the degradation of the physical habitat supporting the species or habitat of interest, (iii) fragmentation and connectivity within and between areas, (iv) spatial extent, distribution and condition of the species and habitats before recovery, (v) autecological factors such as fecundity, dispersal, growth and mortality, and (vi) biogeographic changes in species and habitat distribution.

These factors are largely site-specific and the MarLIN assessments of the recovery potential of habitats and associated species assemblages caution that recovery is not a deterministic process that can be readily predicted. MarLIN provides a generic caveat for recovery assessments that:

“the resilience and the ability to recover from human induced pressures is a combination of the environmental conditions of the site, the frequency (repeated disturbances versus a one-off event) and the intensity of the disturbance. Recovery of impacted populations will always be mediated by stochastic events and processes acting over different scales including, but not limited to, local habitat conditions, further impacts, and processes such as larval-supply and recruitment between populations.”

Notwithstanding these caveats, the MarLIN project and other studies have assessed the recovery potential of features (habitats and species), with life history traits of associated species and the habitat type being of particular predictive value (MES, 2008, Borja and others 2010, Duarte and others 2015, Kaiser and others, 2006). Meta-analyses of over 100 fishing impact manipulations show that slow-growing, sessile species such as bivalves, sponges and soft corals take much longer to recover (up to 8 years) than mobile biota with shorter life-spans such as polychaetes and malacostracans (<1 year) (Kaiser and others, 2006, Sciberras and others 2018).

Habitat processes and characteristics are also a key factor determining recovery rates. For sand and coarse sediment habitats that are dominated by physical processes in areas of high wave action or water currents, habitat restoration is typically relatively rapid (days to a few months), whereas more sheltered muddy sand and mixed habitats that are mediated by a combination of physical, chemical and biological processes, habitat restoration is much longer (months or >1 year, Dernie and others 2003). For habitats characterized by long-lived habitat forming species, recovery may require longer timescales or for particularly sensitive features, recovery may not occur. Deep-sea corals and sponges grow more slowly and recovery times from trawling disturbance or oil spills may range from 30 years to more than a century (Duarte and others 2020). Persistent pressures or regime shifts may also prevent natural recovery.

Borja and others (2010), reviewed 51 long term case studies where recovery was monitored after cessation of pressures and found that although, in some cases, recovery can take <5 years, especially for the short-lived and high-turnover biological components, full recovery of coastal marine and estuarine ecosystems from over a century of degradation can take a minimum of 15–25 years to attain the original biotic composition and diversity may take longer. The time span of recovery after removal of the pressure was highly variable, extending from several months (in the case of meiofauna) to more than 22 years (in hard-bottom macroalgae and some seagrass species).

Assessing natural recovery potential

The MarLIN project provides information to support marine conservation, management and planning that includes assessments of recovery potential of habitats (based on the UK and EUNIS marine habitat classifications) after a range of pressures have been alleviated. MarLIN recovery potential assessments have been used to provide an assessment of natural recovery potential and are supplied in Appendix 1 and the separate Objective 5 Excel spreadsheet (Decision making framework). The recovery from abrasion pressure was used as the basis of the recovery assessments shown in Appendix 2 as this is typically one of the pressures with the most developed evidence base from fisheries. The MarLIN project does not assess recovery from subsurface penetration or extraction for rock habitats as these habitats are not considered likely to be exposed to these pressures. The recovery assessments for the penetration and extraction pressures are supplied within the Excel spreadsheet that accompanies this report as they were considered to be informative for a higher degree of disturbance for sedimentary habitats. For each soft sediment EUNIS biotope a decision was made whether to use the abrasion or penetration/extraction assessments. Where key species on which sensitivity was based were larger burrowing infauna, the more damaging penetration and extraction pressures were used as more representative of recovery potential.

The habitat components that underpin the recovery assessment for each EUNIS level 5 biotope were identified from MarLIN where recovery potential was assessed as ‘Medium’ (full recovery within 2-10 years), ‘Low’ (full recovery within 10-25 years) or ‘Very low’ (Negligible or prolonged recovery possible; at least 25 years to recover structure and function). For each habitat any further information found within the evidence review was included in the Excel spreadsheet (column AQ). For some species, additional searches were undertaken or the BIOTIC database of life history traits was used to supplement information (see Appendix 1 for literature search terms).

EUNIS level 5 habitats that are assessed as having ‘Very low’ recovery, either occur on soft rock or in seeps and vent habitats where the habitat is not predicted to recover, or are biogenic habitats formed by slow-growing species such as maerl, horse mussels, cold-water coral reefs and *Serpula vermicularis* reefs. Slow or very slow recovering species within habitats that determine the recovery potential include long-lived slow growing Axinellid sponges, sea pens and anthozoans.

Muddy sands and deeper more stable circalittoral reefs, muds and sands and biogenic habitats of blue mussel (*Mytilus edulis*) and the Ross worm (*Sabellaria spinulosa*) are expected to have ‘Medium’ recovery potential. Recovery is predicted for these species based largely on their life history traits including longevity and larval dispersal potential of the species assemblage. There are typically evidence gaps for small infaunal components around ecology, life history traits and, hence, recovery potential.

Habitats with ‘High’ recovery occur mainly within environments subject to disturbance that are dominated by physical processes and are characterized by smaller, robust, fast-growing, short-lived species. Examples include sediments that occur in variable salinity and those that are very tide-swept or are found in surge gullies.

Most of the evidence for recovery from disturbances is based on fishing activities, dredging and aggregate extraction. These activities mainly affect soft-sediment habitats and there is, therefore, less direct recovery evidence for rock habitats, particularly animal dominated circalittoral rock habitats.

Objective 3 Assisted recovery options

The literature review (see Appendix 1 for search terms) undertaken for this project identified a number of measures that are relevant to recovery. Mitigation is included briefly here as it supports recovery by minimising impacts on species, habitats, and ecosystems. Mitigation measures (ABPmer, 2020), used before, during or after projects include measures that:

- minimise disturbance, loss of habitat, impacts on migration pathways or sensitive sites by considering site selection and project scale;
- minimise disturbance to sensitive locations with exclusion zones;
- minimise disturbance to benthic habitats though infrastructure design;
- minimise habitat disturbance during decommissioning with the use of Best Practicable Environmental Option decommissioning standards;
- minimise impacts on hydrographic factors and the sedimentary regime through design of infrastructure, demonstrated through appropriate modelling;
- enhancement of infrastructure to increase species diversity;
- minimise impacts on habitats through micro-siting;
- minimise release of suspended sediments in the water with construction techniques;
- minimise impacts of construction on sediment processes with construction techniques;
- minimise impacts to physical processes by undertaking work or operating at appropriate tidal states, and
- consideration of the introduction of invasive non-native species and development of a Biosecurity Management Plan.

Restoration actions differ in terms of their applied strategies and approaches and whether they are focused at population or habitat levels or at an even broader landscape scale: the ecosystem level (for example, restoration of a watershed ecosystem). The focus of the evidence review was assisted recovery options, which according to Elliott and others (2007) have been divided into eco-engineering options that improve physico-chemical factors and processes (including sediments, water quality and quantity) and those that engineer the ecology, by replanting or restocking species. In practice both approaches may be required for recovery to occur. For instance, if the water conditions are detrimental to the introduction and persistence of the species, the water quality itself should be restored prior to implementing any restoration actions. Ecological approaches may also improve conditions. Examples include restocking filter feeding bivalves to improve water quality, an approach that has been used in small, enclosed dock habitats (Hawkins and others, 2020).

Systems displaying alternative stable states may also need both physical and biological forcing to return to the desired state (Scheffer and others, 2001). Physical–chemical improvements can alter bottom-up forcing to support restoration of habitats but putting back in place top-down control, for example, grazers will require biological intervention (Nystrom and others, 2012).

Some general principles apparent across studies were summarized by Geist and Hawkins, (2016). In more enclosed systems, these include:

- stopping as many impacts as possible and harnessing natural recovery processes;
- action to reinstate the geomorphological template and hydrodynamic processes;
- control of excessive bottom-up forcing such as eutrophication, and
- restoration of top-down control by higher trophic levels concentrating on ecosystem engineers, keystone species, and habitat forming and shaping species or assemblages.

In more open systems such as bays and offshore marine areas the main, feasible restoration suggestion was to remove impacts and rely on natural recovery as far as possible (Geist and

Hawkins, 2016). Steps to support recovery of natural habitats through management of pressures have been widely implemented in the UK. Examples include:

- use of byelaws or other enforcement measures to ensure activities avoid sensitive areas;
- designation of Marine Protected Areas to protect species and habitats of conservation interest, and
- improvements in water quality through reduction in contaminants

The first step in enabling recovery of any degraded ecosystem is to remove or reduce current impacts. Where impacts are not direct and clearly linked to habitat loss or degradation investigation may be required to target actions that will have most effect (Geist and Hawkins, 2016). This step is common to all restoration approaches and is, therefore, not included in Table 2 that considers eco-engineering to improve habitat conditions and Table 3 that considers approaches to improve ecology by restoring species populations.

Further information is provided for assisted recovery options that were considered relevant to the UK in Appendices 3-11 of this report.

Table 2. Examples of assisted recovery using direct interventions or ecoengineering that improves the physico-chemical processes, including water quality and quantity and habitat

Component	Measures	References	Appendix
Habitat: Shallow subtidal mudflats	Sedimentation polders using concrete-reinforced brushwood fence lines are often employed to accrete new mudflats in front of new dykes in Netherlands and Germany.	ABPmer, 2020	No. UK examples not found
Habitat: Sands and gravels	Sediment recovery through shell-seeding, gravel seeding, dredging unwanted material from the seabed. Bed levelling and recontouring, filling of excavation pits	Saunders and others, 2010, Cooper and others, 2011, Cooper and others, 2013	1
Habitat: Subtidal sediment	Targeted sediment placement (Sediment capping)	Cooper and others 2013. Oncken and others 2022	2
Habitat: Boulder reefs	Boulder field restoration. Repositioning boulders to restore boulder reefs	Støttrup and others 2017, Liversage, 2020	3

Component	Measures	References	Appendix
Bivalves	Sediment harrowing to resurface buried shells (ineffective)	Bromley and others, 2016	6
Habitat: Rock reef	Creating reefs by adding reef blocks or artificial hard substratum	Pondella and others 2018, Jensen, 2002	4
Water quality, hydrology: Saline lagoons	Resalination; hydrological modifications, algae harvesting	Ghosh and others, 2006, Thelen and Thiet, 2009, Thiet and others, 2014	No as examples not UK and focussed on larger lagoons.
Predator exclusion	Fences to exclude crabs that predate on blue mussel (*Mytilus edulis*)	Schotanus and others, 2020	5
Reduction wave action	Artificial breakwaters (in the form of small fences in the intertidal)	Schotanus and others, 2020	5

Table 3. Assisted recovery that improves the ecology by repopulation species through improving brood stocks (population structure), restocking, translocation or other habitat interventions.

Species	Measures	References	Appendix
Oysters (*Ostrea edulis*)	Restocking: Transplanting juveniles (seed or spat) from hatcheries or donor populations. Substratum restoration/ enhancement: adding cultch to provide suitable settlement cues and substratum Oyster nurseries (submerged protected cages).	Preston and others 2020, Liversage, 2020	6

Species	Measures	References	Appendix
Blue mussel (*Mytilus edulis*), cockles	Restocking: Transplanting juveniles (seed or spat) from hatcheries or donor populations	Capelle and others, 2014)	5
Horse mussel (*Modiolus modiolus*),	Restocking: Transplanting from donor populations (Experimental technique and not clear beds occur in England's waters).	Fariñas-Franco, and Roberts, 2014	No.
Cockles (*Cerastoderma edule*)	Hatchery restocking	Pronker and others, 2015.	No. Experimental technique not used in the UK.
Lobsters (*Hommarus gammarus*)-	Restocking from hatcheries	Ellis and others, 2015	No. mobile species.
Blue mussel (*Mytilus edulis*)	Artificial substratum, cement based substratum	Mussel, coir nets- de Paoli et al 2015 Wadden Sea; Shotanus and others 2020, Christensen and others 2015	5
Seagrass	Transplantation or translocation of seagrass seedlings, sprigs, shoots, or rhizome	Gamble and others, 2021	7
Seagrass	Seed collection and re-seeding	Marion and Orth, 2010	7
Seagrass	Bags of Seagrass Seeds Line (BoSSLine)	Unsworth and others, 2019	7
Kelp	Transplantation, green gravel, seeding	Fredriksen and others, 2020	8

Species	Measures	References	Appendix
Maerl	Translocation	Sheehan and others 2015	9
Seahorses	Artificial holdfasts (Experimental approach Not used in UK. Report focusses on habitat not species restoration.	Correia and others 2013 - (*Hippocampus guttulatus*) Portugal	No.
Sabellaria alveolata	Translocation (attached to boulders), Artificial reefs	MMO, 2019	No. Artificial habitat not assisted recovery
Gorgonians	Translocation and transplant of fragments caught as by-catch and attached to small stones for placement.	Casoli and others, 2022	No. Technique not tested in UK
Cold water corals	Translocation and transplant of fragments	Linares and others 2020, Montseny and others, 2020	No. Experimental technique.

Effectiveness of natural versus assisted recovery

Lotze and others (2011) reviewed evidence for marine habitat recovery. They found that, in 95% of cases, recovery had occurred directly as a result of the reduction or removal of pressures (associated with human activities) that had led to the degradation of habitats or depletion of species. Pressure removal was most successful when used in combination with other management measures (such as habitat protection, bans or restrictions on certain types of fishing gear, measures to improve water quality, reintroduction of species, active habitat restoration and active protection of breeding colonies) whilst management measures alone, that did not include direct removal of pressures, led to recovery in 72% of cases. This clearly indicates the importance of management where anthropogenic disturbance is the primary cause of species and habitat loss. Removal of pressures for the majority of habitats is likely to be the most effective approach to supporting recovery, where the habitat is able to recover and support a similar biological assemblage that can recolonise (through migration or propagule supply).

The effectiveness of assisted recovery versus natural recovery is apparent for habitats that have undergone declines in extent and distribution and that will not recover, or are unlikely to recover, without assisted recovery measures. Examples of habitats that are recovering only with active intervention are boulder reefs that have been removed in their entirety (see Appendix 5) and oysters and seagrass beds which have suffered extensive, historical declines and suffer from low natural recruitment due to multiple factors including changes in habitat quality, negative feedbacks (turbidity and water movements) and lack of propagules inhibiting recovery (see Appendices 8 and 9). For habitats comprising long-lived species, assisted recovery may require the same timescale as natural recovery in order to establish population age structure and typical features such as canopy structure (for example for *Ascophyllum nodosum*) and biogenic reef structures (for example *Lophelia pertusa*).

Translocation of species of conservation interest

Translocation (relocation) or transplanting of species of conservation interest is widely practised in terrestrial and freshwater environments, for example for the protected great crested newt. In the marine environment such practices are not widespread for species other than blue mussel (commercial relaying) and for active habitat restoration (bivalves and seagrass). Hiscock and others (2013) identified a number of rare species that could be translocated. These included the lagoonal worm *Armandia cirrosa* and the spiny crawfish *Palinurus elephas*. Translocation to assist recovery may only be appropriate for a very few marine species, such as those that have become locally extinct and are unlikely to recolonize an area through their own dispersal mechanisms. Attached species are less suitable unless boulders and cobbles could be moved. The translocation of fragments of species that regenerate easily such as sponges and corals have been trialled but only for tropical or Mediterranean species (Casoli and others, 2022).

Artificial habitats and assisted recovery

Species have been recorded as settling in artificial habitats, especially artificial reefs created for a range of purposes. Examples include settlement of *Sabellaria spinulosa* on artificial reefs (Almeida and others 2016). These examples have been noted in the Objective 5 framework Excel spreadsheet. However, settlement on artificial habitats is not considered to be an example of assisted recovery where the habitat itself is markedly different. For the habitat A5.611 (*Sabellaria spinulosa* on stable circalittoral mixed sediment), artificial hard reef would not be considered to represent this sedimentary biotope. *S. spinulosa* also occurs on hard rock (A4.221 *Sabellaria spinulosa* encrusted circalittoral rock), however, the effectiveness of artificial reefs as an assisted recovery approach is debatable as colonization may be highly variable unless species are manually transplanted (approach trialled for kelps, see Appendix 10). It is unlikely that artificial reefs or natural rock would be placed in the environment to restore this habitat given the uncertainties that this would lead to the restoration of a *S. spinulosa* reef.

Implementation of assisted recovery

The most widespread approach to assisted recovery in the marine environment is the management of pressures that lead to impacts on habitats and species. Eco-engineering approaches such as measures to enhance water quality and reduce sediment contamination have occurred across broad areas and have supported recovery. To date, assisted recovery efforts that engineer the ecology have focussed on relatively large, sedentary species that form biogenic habitats. These include bivalves that form reefs or beds (oysters, blue mussels) and seagrasses. These biogenic habitats support high levels of ecosystem service provision, and goods and benefits such as maintaining the populations of commercially exploited species, through direct provision of brood stock supply to regenerate populations or nursery functions, for example providing shelter for juvenile fish (Lefcheck and others, 2019). Once reintroduced, these biogenic species can provide fundamental ecosystem functions and processes that will ultimately benefit other associated organisms and support overall system recovery (Powers and Boyer 2014).

Objective 4 Review of assisted recovery options

Information on advantages, disadvantages, costs, benefits, feasibility and risks, challenges and uncertainties of the key assisted recovery approaches identified in Objective 3, are provided in Appendices 3-11 and the results and additional information is summarised here.

Feasibility

For each approach, information on feasibility is provided in Appendices 4-11. Most approaches to assist recovery of marine species can still be regarded as under development in terms of application to English subtidal marine habitats. Application of most approaches is largely experimental and small-scale although seagrass and oyster restoration projects are beginning to be implemented in larger areas (see Appendices 8 and 9).

A key aspect of feasibility is identifying sites where assisted recovery may be successful, particularly when creating biogenic habitats that may have been absent for a long-time. Feasibility investigations require a range of assessments, including environmental conditions and habitat suitability evaluations. Where transplantation is used, donor populations should be matched as closely as possible to conditions in the transplant site. Genetic considerations are also important for species populations. Assisted recovery may provide an opportunity to increase genetic diversity, for example, to introduce populations likely to be more resilient to climate change.

Removal of pressures and assisting recovery over larger spatial scales has been linked to increased recovery success for seagrass beds (Appendix 9). Recovery options that are applicable to small-scales only are unlikely to overcome negative feedbacks such as sediment instability that inhibit recovery. A key example of this are mussel restoration efforts in intertidal flats in the Wadden Sea (see Appendix 7). The small-scale of approaches means that beds cannot establish in dynamic areas, as the wave dampening functions of beds have been lost following historic declines.

As knowledge and experience of overcoming limiting factors and experience increases, assisted recovery feasibility is likely to improve. Cost-effective approaches that can be used over wider areas are being trialled, such as green gravel for kelps (Appendix 10) and seed bags for seagrass (Appendix 9).

Costs associated with assisted recovery approaches.

Assessing the costs of marine restoration consistently across methods and habitats is challenging (Danovaro and others, 2021). Many studies do not provide information on costs and for those that do cost reporting is inconsistent. Evaluating reported costs between countries where labour costs, for example, may vary and across different years (or decades) and currencies is also subject to uncertainty. To allow cost comparisons, costs were standardised to hectares. Many of the costs reported are for small-scale experimental studies and it is not clear how these may scale-up across larger scale restoration activities where costs per unit area may be lowered. These caveats should be recognised for the reported costs in Table 5 below and the cost evaluation presented in Objective 5.

Costing exercises rarely include the research costs to underpin decisions such as choice of appropriate techniques and feasibility studies including pilot studies to support selection of donor and restoration sites, and techniques. Further, the costs of non-consumable laboratory equipment (e.g. stereomicroscopes or diving materials) needed tend not to be included in estimates as pre-existing facilities are usually utilised. Costing exercises do not include the research and background knowledge that is crucial to planning and implementing optimal restoration techniques (Layton and others, 2020; Campbell and others, 2014).

Assisted recovery approaches are typically resource intensive. Støttrup and others, (2017) identified best practice around boulder reef restoration. Best practice tasks with some adaptations and additions are shown in Table 4. Many of these tasks are common to most marine subtidal restoration projects and, given the number and variety, indicate why assisted recovery approaches are resource intensive. Many costings available do not include all these stages, particularly the capital costs around restoration planning and sourcing of information.

For projects that require transplanting individuals (for example seagrass) or labour intensive practices such as hand removal of predators) labour costs are a large part of the overall project costs. Hence, community or volunteer based marine restoration projects usually have lower costs.

Table 4. Tasks required for best practice assisted recovery projects

Project stage	Task description
Capital costs	Public Involvement. Map stakeholders (for example, benefit, affected, manage, interest). Gather local knowledge and information about local interest in the area (for example, harbours) and activities that may need to be managed, cultural heritage in the vicinity
Capital costs	Identify and source scientific inputs needed
Capital costs	Identify activities needed to assist recovery
Capital costs	Identify and obtain information needed. For example, assess feasibility. Information about local hydrodynamics, sediment transport, ecology and biology, other protected areas in the vicinity and possible impacts
Capital costs	Define objectives: biological or ecological (for example, biodiversity baselines, specific biogenic habitats)
Capital costs	Alleviate existing pressures
Capital costs	Assess risks, for example disease prevalence, presence of or risk of introducing invasive non-native species
Capital costs	Design restoration, for example, size, extent, seasonal effects, design and materials

Project stage	Task description
Capital costs	Prepare a management plan
Capital costs	Obtain permits or licences for feasibility studies, implementation and monitoring
Operation costs	Communication with stakeholders to minimize problems
Operation costs	Manage risks for example mark working areas with buoys, manage biosecurity to limit introduction or spread if invasive non-native species
Operation costs	Obtain and deploy required materials for example: oyster nurseries, spat collection, seed collection, other broodstock, boulders, reef materials
Maintenance costs	Decide which activities to allow, to what degree and how to manage these activities (tourist, fishery, diving, navigation, etc).
Maintenance costs	Monitor against baselines and objectives
Maintenance costs	Protection of site, for example from unlicensed fisheries, poaching

Assessing assisted recovery costs

Bayraktarov *and others* (2016) categorised groups of costs for various marine habitat restoration projects including planning, land acquisition, construction, financing, maintenance, monitoring and equipment repair and replacement as factors which affect the cost, feasibility and likelihood of success of marine restoration. They synthesized 235 studies with 954 observations from restoration or rehabilitation projects of coral reefs, seagrass, mangroves, salt-marshes, and oyster reefs worldwide, and evaluated cost, survival of restored organisms, project duration, area, and techniques applied. The majority of restoration projects were short-lived and seldom reported monitoring costs. Restoration success depended primarily on the ecosystem, site selection, and techniques applied rather than on money spent. Findings showed that while the median and average reported costs for restoration of one hectare of marine coastal habitat were around US $80,000 (2010, £59,600) and US$1 600, 000 (2010, £ 443,745), respectively, the real total costs (median) are likely to be two to four times higher.

For this project, cost assessments have been divided, where possible into capital costs (cost for planning, purchasing, construction, and financing) and operation and maintenance costs. While there are many gaps in the evidence, available information is supplied in Appendices 3-11).

Information on specific costs for assisted recovery options are provided below in Table 5. These have been taken from a wide range of sources and converted to GBP as detailed in the table. Identifying costs was challenging as these tend not to be reported or, if reported, are not broken down to tasks. Restoration costs per hectare varied between £10,600 for dredging of subtidal sands and gravels to just over £1.5 million for total project costs for boulder restoration.

Table 5. Restoration project costs. See appendices for further information. All costs were rounded to nearest £100. Further details are provided in Appendices 3-11. Spatial and temporal scale refers to project spatial scales (Not applicable where mean costs from a range of projects are reported). Costs were standardised to hectares (ha).

Technique	Spatial scale*	Temporal scale	Costs £/ha	Reference and notes
Renville bay Galway Oyster reef (Cost for first year including materials, staff etc)	0.642/ha	First year costs	£323,400	Hynes and others 2022, costs in Euros (conversion to 2022, GBP)
Shell seeding for sediment restoration (based on materials only)	0.34km^2	One-off initial cost	£154,400	Hynes and others 2022 from Essex Native Oyster Initiative
Sand and gravel restoration: Dredging (including licensing, baseline and post restoration surveys)	0.34km^2	One-off initial cost	£10,600	Cooper and others 2013. Costs converted to 2022 value (see Proforma 1)
Sand and gravel restoration: Capping gravel seeding) (including licensing, baseline and post restoration surveys)	0.34km^2	One-off initial cost	£26,500	Cooper and others 2013. Costs converted to 2022 value (see Appendix 3)
Sand and gravel restoration: Bed levelling (including licensing, baseline and post restoration surveys)	0.34km^2	One-off initial cost	£11,200	Cooper and others 2013. Costs converted to 2022 value (see Appendix 3)
Boulder restoration	27,600m^2 (2.6ha)	Total project costs	£1,546,500	Based on online LIFE funding website, total

Technique	Spatial scale*	Temporal scale	Costs £/ha	Reference and notes
				cost of €4,808,398.
Kelp restoration-seeding	Not applicable	Not applicable	~£285,700-£904,100	Eger and others (2021) ($/ha) see Appendix 10.
Kelp restoration transplantation	Not applicable	Not applicable	£367,900-£918,000/ha	Eger and others 2021; Groeneveld and others, 2019), see Appendix 10
Green gravel	Not applicable	Not applicable	£53,635	Fredriksen and others, 2021
Seagrass (Dale, Pembroke) excluding monitoring (hessian bags)	2 hectares (20,000m^2)	Not reported	£200,000	Cited from Kent and others, 2021
Seagrass (Sweden) shoot transplantation including site selection and evaluation, harvesting, planting and monitoring.	Not reported	10-year cost	£105,000-220,000	Moksnes and others, 2021, cited from Kent and others, 2021. Not converted.
Seagrass (Sweden) shoot transplantation including site selection and evaluation, harvesting, planting and monitoring.	Not reported	10-year cost	£218,692 - £633, 065	Moksnes and others, 2021, cited from Kent and others, 2021. Not converted.
Seagrass (developed) Capital and operating	Not applicable	Not applicable	£516,809	Bayraktarov and others 2016. Converted to

Technique	Spatial scale*	Temporal scale	Costs £/ha	Reference and notes
				GBP and to 2022 prices
*****Oyster reef (developed) Capital and operating**	Not applicable	Not applicable	£634,860	Bayraktarov and others 2016. Converted to GBP and to 2022 prices

Benefits of assisted recovery

The value of restored ecosystem services has been used to assess the benefits from assisted recovery (Basconi and others, 2020). While ecosystem services, goods and benefits are frequently alluded to in reports on assisted recovery, valuing these is more challenging and limited examples were found.

Eger and others (2021) valued the ecosystem services of four major kelp genera (including *Laminaria*) at around $135,200 to $177,100 ha/year (approximately £103,200 to £135,200). The value the habitat provides could potentially offset the costs of kelp restoration within 2-7 years. Restoration of kelp habitats also produces socio-economic benefits through the possible revival of related industries such as fisheries (Claisse and others, 2013; Bertocci and others, 2015).

There can also be social benefits from restoration, including job creation (Edwards and others 2013), increased community engagement, and educational opportunities. Seagrass habitats provide examples of community engagement and citizen science with projects using volunteer snorkelers to collect seeds and volunteers, including school children, to fill bags for reseeding. UK projects have involved extensive work with local stakeholders and communities, who help choose the exact location of planting, to ensure that schemes are widely supported and have a positive benefit (Jones and others, 2018).

Cost-benefit analysis

Providing cost benefit analyses for assisted recovery vs. natural recovery is challenging due to the lack of published costings for assisted recovery and limited assessments of benefits (Danovaro and others, 2021). Examples of cost-benefit analyses were found in the literature but these were limited in extent, detail or application to the scope of this study (English subtidal habitats). In general marine ecosystems supply high-levels of ecosystem services but due to the high costs of marine restoration, the ratio of cost to benefit for marine habitats is typically lower when compared to terrestrial approaches (de Groot and others, 2013). However, Stewart-Sinclair and others (2020), found that benefit values (estimated as the monetary value provided by ecosystem services of the restored habitats) outweighed costs for restoration for mangrove, saltmarsh and coral reef habitats. The cost-benefit ratio of assisted recovery is likely to improve further as assisted recovery options increase in spatial scale and reduction of costs.

Economic analysis of the costs and benefits of oyster restoration in North Carolina, USA, produced expected benefits ranging from $2 to $12 for every dollar invested in terms of enhanced recreational fishing, improved water quality, and commercial fishing (Callihan and others, 2016). While Eger and others, (2021) found that the benefits provided by kelp habitat could potentially offset the costs of kelp restoration within 2-7 years.

The cost-benefit of possible techniques to restore the physical properties of the seabed following aggregate dredging and their likelihood of success was assessed by Cooper and others (2013). As part of this analysis, the ecosystem service benefits of sands and gravels were valued at an impacted (dredge site) and an unimpacted reference site. The value of carbon sequestration ranged from £58.50 to £148.29 per km^2 to £6.79 to £46.30 per km^2. The value at the non-impacted (reference) site ranged between £81.13 and £186.69 per km^2. An analysis of the ecosystem services and goods/benefits produced by the site was used to determine whether intervention was justified and it was concluded that for this site the costs outweighed the benefits.

Hynes and others (2022) estimated the cost and benefit values associated with a coastal walking trail in Western Ireland and its protection from climate related events using either hard engineering solutions or through the restoration of a protective oyster reef bar. After calculating the annual recreational benefit value associated with the coastal walking trail and the costs of the alternative approaches to its protection from storm surges, a cost–benefit analysis was carried out with projects over a 20 year time period. The annual net recreational benefit was valued at €642,063 (approximately £54,000). The initial capital cost for the estimated 1070 m of coastal protection was estimated as between €1,092,763 and €12,369,618 (£916,691 and£10,376,561) for permeable rock revetment or impermeable seawalls. The total first year cost of establishing the reef was €259,796 (£217,936). The oyster reef nature based solution had the lowest cost and was therefore the more attractive option from an economic perspective. The paper did not assess the value of other ecosystem services or goods and benefits provided by the reef and the benefits considered from both the engineered and the oyster reef options –coastal protection- were the same.

Risks, challenges and uncertainties

A number of risks, challenges and uncertainties have been identified for the assisted recovery approaches in the Appendices (3-11). Some commonalities are apparent and are outlined below.

Risks

Assisted recovery approaches may impact donor populations (where stock or transplants are obtained, for example seagrass). Consideration should also be given to habitats and species that occur at sites where recovery projects are planned. Restoration activities should be carefully evaluated where they may impact other species or habitats of socio-economic value or conservation importance. Such assessments, licensing and permitting form part of project costs.

Biosecurity risks around introduction or spread of pathogens and invasive non-native species are important for a range of projects that involve the movement of stock (transplantation and translocation) or infrastructure such as artificial beds or that involve vessels. If a restoration site has high impact Invasive Non-Native Species (INNS) present, it is crucial that the local prevalence and impacts on features are understood at the stage of site assessment. Non-native species, even if they are not high impact and invasive, may still need to be considered in project planning and biosecurity measures. The risk of potential spread of such species, for example through the movement of equipment, should be managed (Zu Ermgassen and others, 2020).

The creation of permanent or temporary infrastructure at sea will affect activities and other users and may impact safety. Examples include concerns that boulder deposits in shallow areas may impede shipping by lowering depth (Støttrup and others, 2017, see Appendix 5). Where possible, practitioners should seek to mitigate risks at the site selection stage. A full risk assessment of potential safety concerns should be undertaken, and consideration should be given as to whether the location of the proposed restoration can be altered to mitigate any identified risks.

Challenges

Restoration projects should include costed stages that assess site suitability to ensure local conditions will not inhibit recovery even if initial assessments indicate suitability. This should involve reviewing historic data, making use of habitat suitability models, and monitoring physical and biological parameters in the area that has been selected (Kent and others, 2021).

A key challenge for assisted recovery approaches is the establishment of eco-engineering species. Where these are established they provide positive feedbacks that encourage recruitment and maintain populations. Biogenic habitat forming species such as kelp, seagrass and bivalves, stabilize and trap sediments and dampen wave energy facilitating retention of larvae and juveniles. Where these are lost, translocated individuals are typically either too small, too sparse or too unstable to modify their environment and establish such self-facilitating feedbacks. These limitations may be addressed by:

- Reducing environmental stressors
- Removing or excluding predators
- Planting density
- Selection of optimal habitats

Uncertainties

The scale of assisted recovery projects has been limited by costs and feasibility of techniques. There are key uncertainties around the costs, scalability and success of approaches as outlined in Appendices 4-11.

Collaboration and stakeholder participation is critical to the success of assisted recovery projects

The success of restoration projects has been strongly linked to Interdisciplinary and inter-organizational collaboration (Saunders and others, 2020). Strong local involvement and support from the local community is known to be a key factor contributing to marine conservation success (Saunders and others, 2020). The governance perspective includes understanding the interactions and interdependencies of multiple authorities and competing maritime activities (with different economic, political, social, and cultural interests), all of which operate at different governance levels, ranging from sub-national (coastal governments) to the international arena (Ounanian and others, 2018). Commitment of funding and time contributes to long-term monitoring and management, with substantial levels of government funding in particular associated with marine restoration success (Saunders and others, 2020).

Stakeholder participation allows all partners to actively participate in the process of developing plans and projects, including policy options, before decisions are made. Stakeholder engagement is more likely to be required for inshore areas where multiple activities occur and where there may be a range of interested parties (recreation, commercial, management, scientific, natural history etc. Assisted recovery projects may benefit from engaging stakeholders in the following ways:

1) obtain local knowledge that improves the project design;
2) support for on-going management, voluntary codes, access agreements;
3) obtain stakeholder support for field work or data collection (i.e. citizen science); and
4) stakeholders may gain ownership of the project, ensuring the long-term viability of the restored area.

For example, an advisory board was established for the Kattegat boulder restoration project (Støttrup and others, 2017) with representatives of regional and local environmental management and NGO organizations dealing with fishery, nature protection, diving, tourism and yachting. Apart from the advisory board meetings, public meetings took place to support dialog with the local community. At the meetings, the public were informed about the planning, progress and the results of the project.

Objective 5 Decision support framework

Examples of previous approaches were identified to support the development of a decision making framework around restoration. The sources considered included general references on marine spatial planning with regard to restoration (assisted recovery). These identified a number of commonalities around projects. The list below is not exhaustive but common factors include:

- the need to determine impact to identify the nature, degree, and extent of any injuries to natural resources and services; successful projects consider the specific environmental and ecological context of the restoration site to address the specific conditions that have led to degradation (Saunders and others, 2020);
- the need to alleviate existing pressures that are causing impacts or preventing recovery;
- the need to identify baselines against which to reference recovery or the recovered end state for example, the recovered reference range (Mazik and others 2015);
- the need to identify goals for restoration projects;
- the requirement to identify natural recovery potential and assisted recovery approaches;
- that restoration should target areas in which its feasibility has been assessed (for example, through ecological modelling) and especially where the environmental conditions are suitable for the survival of the target species;
- spatial and temporal scale and sufficient connectivity to source populations should be considered (Saunders and others, 2020);
- restoration approach effects on existing habitats should be considered and future environmental changes should be evaluated, including continued persistence of restored marine habitats subject to climate change (Saunders and others, 2020);
- being positioned to respond quickly to unforeseen events that may prevent the achievement of restoration goals (Saunders and others, 2020);
- monitoring should be in place to evaluate recovery and progress towards a recovered state,
- recovery can be very slow in long-lived, late maturing low fecundity species, so it is necessary to consider appropriate timescales to assess the effects of restoration (Thom and Weliman, 1996). and
- engagement with stakeholders, community involvement, funding and governance.

Decision making frameworks

The range of considerations that restoration projects should consider are diverse, as listed above. Decision making frameworks and tools are important to provide guidance, particularly when considering multiple habitats and/or options.

Saunders and others (2010) developed a framework for decision making around aggregate site restoration that includes the following components. The specific steps in the framework were:

- Step 1: Identifying the need for restoration measures;
- Step 2: Identifying the baseline;
- Step 3: Identifying and screening potential restoration options;
- Step 4: Assessing the positive and negative impacts of restoration options;
- Step 5: Identifying which restoration options are required.

An example of a detailed framework for identifying the potential recovery for MPA features was developed for NatureScot by Mazik and others (2015). The framework components are considered here in terms of assessing recovery potential as part of a decision making framework. The questions they identified around the decision making process were:

- What is the recovery objective – what is the recovered state trying to be achieved?
- What management actions are required during and after recovery?
- What is the underlying cause of decline of a species, community, or habitat?
- To what spatial scale does the recovery objective apply?
- What environmental and ecological conditions are required for a species or habitat to recover?
- What factors are restricting or stopping recovery from occurring?
- What is the cost-effectiveness of recovery balancing the management measures and the resultant ecological structure and function?

The above frameworks were used as the basis for the development of generic high-level considerations regarding recovery potential as well as the specific considerations for individual features that follow.

Suggested framework for decision making:

Many of the decisions around restoration projects and the specific actions to be undertaken are site-specific and cannot be accounted for in a generic framework. However Step 3 and Step 4 of the aggregate restoration framework are more general and similar to Stage 4 and Stage 5 of the framework developed by Mazik and others (2015).

We suggest that the framework to support advice and management options for Natural England should be part of a more generalized decision making workflow outlined below (see Figure 1).

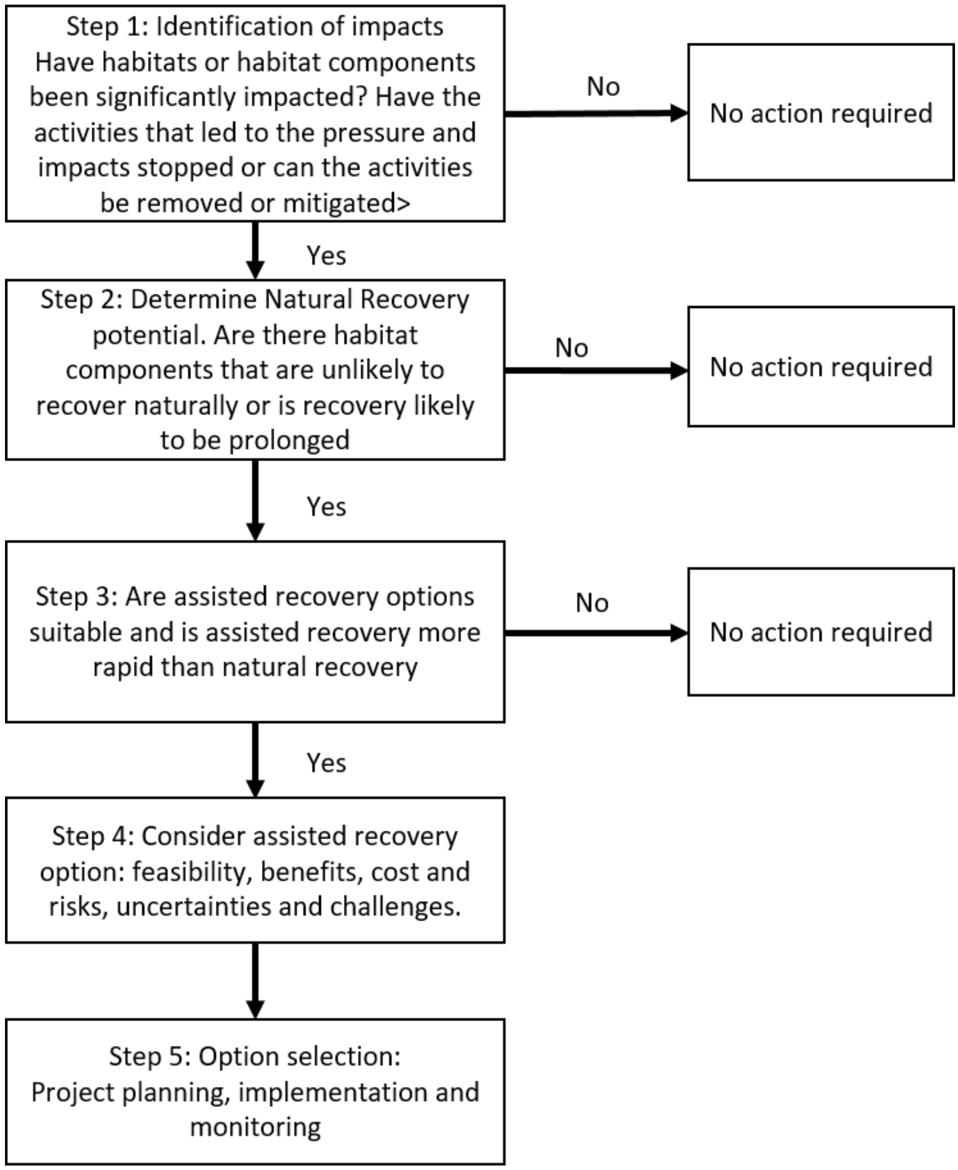

Figure 1 Outline decision making flowchart illustrating key steps to assessing natural recovery potential vs assisted recover options.

Step 1: Identification of impacts (site-specific)

Have habitats or habitat components been significantly impacted? This question requires knowledge of a baseline, that is, the state or condition of the habitat (or habitat feature) prior to impact. Stage 1 of Mazik and others (2015) provide examples of detailed information capture regarding impacts on features and Saunders and others (2010) provides examples of assessing ecosystem structure and function that could be adapted. A key question is whether the pressures that cause these impacts can be removed or mitigated (See Objective 3 for examples of mitigation options). If it is not possible to remove impacting activities that are on-going or to provide mitigation then the impacts will continue and natural recovery or active recovery will be prevented.

Step 2: Determine natural recovery potential.

Objective 2 provides an overview of recovery potential, however this will be mediated by site-specific factors such as patch-size, connectivity to source populations, presence of suitable habitat etc., so this consideration is partially site specific. Step 2 should identify the natural recovery potential and identify slower recovering components or those components that are not expected to recover without intervention. Additional information added to the MarLIN assessment includes information on reproductive mechanisms, dispersal potential, lifespan and generation time. This information is sourced from the traits database BIOTIC[1] and is not available for every species.

Step 3: Are assisted recovery options suitable and is assisted recovery more rapid than natural recovery?

Objective 3 identifies assisted recovery options with more details provided in Appendices 3-11 for each restoration technique. Based on the identification of impacts in Step 1 and Step 2, assisted recovery options that address these singly or in combination, for example changes to hydrology, sediment, water quality and recovery of species or species assemblages should be identified.

Step 4: Consider assisted recovery option- feasibility, benefits, costs and risks

Detailed consideration should be given to feasibility, costs and benefits and any risks uncertainties and challenges identified in the Appendices produced for Objective 4. Feasibility, benefits and costs can be scored-as shown below; feasibility (Table 6), cost (Table 7) and benefits (Table 8). Risks, uncertainties and challenges should be considered based on the Proformas provided. The assessment of ecosystem services is based on reported assessments by Potts and others (2014). These assessments score the provision of ecosystem service and confidence in that provision for supporting service.

Table 6. Step 4 Feasibility of approach categories

Category	Description
None	No approaches identified for slow recovering components.
Low	Proposed assisted recovery approaches are experimental in approach. The literature available to support approach and success is limited or success rates are either low or have not been reported.
Moderate	Approach has been trialled at more than one site, the approach is relatively well understood and successful and supported by case studies.

[1] BIOTIC (Biological Traits Information Catalogue – www.marlin.ac.uk/biotic

Category	Description
High	Well-studied and applied approach. Factors underpinning success and failure and methods are understood, restorations effort are typically successful.

Table 7. Costs associated with assisted recovery

Costs-Capital/ha	Costs-Operation/ha	Costs- Maintenance/ha
None	None	None
Low (<£25,000/ha)	Low (<£25,000)	Low (<£25,000)
Moderate (<100,00/ha)	Moderate (<£250,000)	Moderate (<£50,000)
High (>£100,00/ha)	High (>£250,00/ha)	High (>£50,00/ha)

Table 8. Ecosystem services assessed by Potts and others (2014)

Supporting services	Regulating services	Goods and benefits from provisioning services	Goods and Benefits from Regulating services	Goods and Benefits from Cultural services
Primary production Larval / Gamete supply Nutrient cycling Water cycling Formation of species habitat Formation of physical barriers Formation of seascape	Biological control Natural hazard regulation Regulation of water and sediment quality Carbon sequestration	Food Fish feed Fertiliser Ornaments (incl. aquaria) Medicine and blue biotechnology	Healthy climate Prevention of coastal erosion Sea defence Clean water and sediments Imobilisation of pollutants	Tourism / Nature watching Spiritual / Cultural wellbeing Aesthetic benefits Education

Step 5: Option selection: Project planning, implementation and monitoring (site-specific)

To establish if ecosystem restoration is possible within the desired location(s), this step requires further detailed feasibility studies, site selection processes and the determination of any significant ecological, logistical, legislative or financial barriers to restoration.

Areas selected for restoration techniques that require direct intervention and monitoring must be accessible at important times and at regular intervals throughout the year to conduct maintenance and monitoring activities (if required). Travel time, staff safety, water depth, wave height and storm frequency may all be factors which affect site accessibility both inshore and offshore, as well as the associated costs of access and monitoring.

Applying Steps 2-4 of the Framework to the UK Marine Habitat Classification

The framework was applied to the UK Marine Habitat Classification (UK MHC). The results are provided in the Objective 5 Excel spreadsheet. Each UK MHC/EUNIS record was assessed against recovery (Objective 2), recovery options for each habitat with recovery potential >2 years (Objective 3) and the feasibility, costs and benefits (based on ecosystem services). The section below explains the steps undertaken.

Habitats that were excluded from the framework application.

Habitats that were not recorded in English inshore and offshore waters were excluded. Examples include:

- A3.114 Sparse *Laminaria hyperborea* and dense *Paracentrotus lividus* on exposed infralittoral limestone and
- A4.113 Mixed turf of hydroids and large ascidians with *Swiftia pallida* and *Caryophyllia smithii* on weakly tide-swept circalittoral rock.

Biotopes that recover rapidly (within 2 years) were not included in the assessment framework. For rock habitats, abrasion represents a high level of disturbance to the mainly epifaunal communities that characterize these habitats. For the infaunal assemblages of soft sediment habitats, abrasion on the surface may not cause significant disturbance. Therefore, for all sedimentary habitats the recovery times to penetration and extraction pressures were also considered to indicate natural recovery times. Therefore, for all rock habitats, no EUNIS records at Level 3, 4 or 5 with high recovery to abrasion were considered in the framework. For sedimentary habitats, the assessments for penetration and extraction were considered when deciding if high natural recovery rates were considered appropriate. The records excluded from further assessment based on high natural recovery are labelled within the assessment framework cells as, ‘Not assessed: high recovery’.

Habitats at EUNIS Level 2 and EUNIS Level 6 were not included in the assessment as these were considered too broad (Level 2) or too detailed and to replicate information at EUNIS Level 4 or 5 (for EUNIS Level 6). Level 6 biotopes were checked for recovery times that differ from the Level 5 or 6 records as these are based on distinct species or habitats that could be included in the framework. Examples of these include the slow recovery soft rock biotope ‘A3.2113 *Laminaria digitata* and piddocks on sublittoral fringe soft rock’.

Habitats characterized by invasive non-native species were not considered candidates for assisted recovery and were assessed as ‘Not assessed; habitat recovery high and characterized by invasive non-native species’. Habitats within this category are:

- A3.15 *Sargassum muticum* on shallow slightly tide-swept infralittoral mixed substrata, and
- A5.422 *Crepidula fornicata* and *Mediomastus fragilis* in variable salinity infralittoral mixed sediment

Artificial habitats, such as A4.72 Circalittoral fouling faunal communities, were also excluded. These were labeled as ‘Not assessed: artificial’.

Applying Step 2 Determine natural recovery potential

Natural recovery potential was identified as outlined in Objective 2. For each EUNIS record, recovery was scored, typically based on MarLIN abrasion scores. When the physical habitat has been removed for rock and stone reefs and sediments recovery is typically prolonged and depends on sediment/rock supply (for example boulder movement by storms). Natural recovery of mobile and non-mobile sediments was assessed as very low. Habitat recovery was typically presented in the table at EUNIS level 4, while EUNIS level 5 records were completed to show recovery of the biological assemblage.

Applying Step 3

Assisted recovery options may still require time for population recovery to a typical age structure and therefore, the assisted recovery timescales presented in the Objective 5 Excel spreadsheet are generally the same as the natural recovery potential. Given the high costs and resources required for assisted recovery options it is likely these would only be considered where pressures have been removed but populations are unlikely to recover naturally due to loss of connectivity, changes in habitat conditions (for example, negative feedbacks).

Applying Step 4 Feasibility

Feasibility of restoration options was assessed using the categories in Table 6 above. Assisted recovery approaches to support habitat recovery and repopulate species have been largely experimental and small-scale. Hence, for most approaches, feasibility was assessed as low. The exception was restoration of reef habitats as this practice has been widely undertaken in the marine environment although usually around creating infrastructure.

Applying Step 4 Costs

The costs presented in Table 9 were used to assign the costings to support the feasibility assessments presented in the Objective 5 Excel spreadsheet. For all approaches there are uncertainties around the applicability and consistency of costings across approaches and the scaling to hectares (see discussion). The costings should therefore be treated cautiously. Where there was no evidence, characteristics of the approach were used to assign a cost category as outlined in Table 6 (above).

Table 9. Where cost information was unavailable the following approach characteristics were used to guide costings to support feasibility assessments.

Costs	Capital	Operation	Maintenance
None	No examples	No examples	Operation involves a single action with no on-going maintenance required
Low	Approaches that are offshore standard operations, requiring no infrastructure or living material and less likely to require thorough feasibility studies	Approaches that require offshore standard operations, requiring no infrastructure or living material and one-off interventions e.g. bed levelling	Likely for offshore standard operations, requiring no infrastructure or living material and one-off interventions e.g. bed levelling
Moderate	Approaches that are inshore operations, that involve living material and hence feasibility studies, licensing and biosecurity	Approaches that require relatively simple operations without complex infrastructure, labour intensive actions and that are relatively short-term	Likely to require some maintenance or monitoring to support recovery, for example, predator/grazer or competitor removal.
High	Approaches that require permanent or complex infrastructure, hatcheries and/or likely to require extensive consultation, planning or where extensive or complex feasibility testing is required	Approaches that require complex operations with complex infrastructure or support from hatcheries, labour intensive actions and that are relatively long-term. For example hand transplanting vs broadcast seeding.	Likely to require on-going intensive maintenance to support recovery.

Step 4 Assessing Benefits

The Objective 5 Excel spreadsheet provides the ecosystem service and goods and benefits scoring associated with habitats from Potts and others (2014). To provide an overview and prevent double counting between services and associate goods and benefits, just the goods and benefits scores were summed and presented in the final column of the table. These scores provide a high-level summary indication of the level of benefits but are not economic values. Benefits were scored as Low (≤10), moderate (11-19) and High (>20).

Results summary

The table below (Table 10) presents an overview of the feasibility, costs and benefits of recovered habitats for each assessed approach, while the Objective 5 Excel spreadsheet provides an audit of scores. The results indicate that across approaches feasibility is low, costs for approaches to assist recovery of species populations are high but that benefits are also high (in the form of goods and benefits delivered by the recovered habitat). The eco-engineering approaches for sands and gravels are cheaper but these habitats deliver fewer benefits.

Marine assisted recovery options focussed on species are expensive and labour intensive although approaches are being developed such as the hessian bag planting method for seagrass seeds and the green gravel approach for kelps, that are lower in cost and scaleable to larger areas. Nevertheless, for most marine species, including larger burrowing infauna (holothurians, large polychaetes) and epifaunal species such as seapens and faunal communities of rock no options are available to support recovery. It is therefore likely that in most cases removal of pressures to support the recovery of degraded habitats and management and conservation of remaining habitats will be prioritised over assisted recovery.

Where pressures are alleviated, assisted recovery is unlikely to be required for habitats with medium-high recovery which are widely distributed, well connected through larval dispersal and characterised by species with short-to- medium longevity. Such habitats are likely to recover rapidly from disturbances where habitat conditions support this. Active recovery is not cost effective for habitats with common species that are likely to recover rapidly (depending on larval supply, connectivity and maintenance of habitats) and that in addition have lower value in terms of ecosystem function, ecosystem services and goods and benefits. Examples of such taxa include crustaceans such as barnacles, amphipods, especially those that occur in disturbed habitats for example, mobile amphipods and small bivalves and gastropods.

Table 10. Assessment of feasibility, costs and benefits from the assessed approaches.

Recovery option	Feasibility	Cost-Capital	Cost-Operation	Cost-Maintenance	Benefits
Natural rock reef	High	High	High	None	Mod-High
Shell seeding	Low	Low (based on gravel seeding)	Moderate	None	Moderate
Gravel seeding	Low	Low	Low	None	Low

Recovery option	Feasibility	Cost-Capital	Cost-Operation	Cost-Maintenance	Benefits
Dredging	Low	Low	Low	None	Low
Bed levelling	Low	Low	Low	None	Low
Maerl translocation	Low	Low	Low	None	High
Kelp-transplanting	Low	Moderate	Moderate	No evidence	High
Kelp-seeding	Low	Moderate	Moderate	No evidence	High
Kelp-green gravel	Low	Moderate	Moderate	No evidence	High
Blue mussel artificial substrate	Low	Low	Low	Moderate	High
Blue mussel relaying	Low	Low	Low	Moderate	High
Seagrass seed restoration (hessian bags) and shoot transplant	Low	Moderate	Moderate	Low	High
Native oyster	Moderate	Moderate	Moderate	High	High

Discussion

Summary of findings

Objective 1: Defining Recovery

The first objective of the project was to review and define recovery. This project uses the terms natural or passive recovery to describe the potential of a habitat and/or species assemblage to move towards a recovered state following the removal of pressures Active or assisted recovery, refers to the application of interventions or measures to initiate or maintain recovery towards but not necessarily to complete the transition to a recovered state. The inherent ability of impacted communities to recover following the removal of pressures is referred to as recovery potential. However, it is recognised that for marine habitats, determining an unimpacted state against which to measure recovery trajectory and the recovered end point may be limited by evidence. Despite this there is clear evidence for some locations and habitats of impacts to condition and for the loss of habitats. This is particularly apparent for biogenic habitats (oyster reefs, and seagrasses) where there have been extensive historic losses resulting in bare, unvegetated sediments.

Objective 2: Subtidal habitats and species recovery potential

The natural recovery potential of habitats and associated species assemblages was assessed, using the MarLIN reported recovery rates to abrasion. Habitats and species with the lowest natural recovery potential either occur on soft rock or in seeps and vent habitats where the habitat is not predicted to recover from physical impacts, or are biogenic habitats formed by slow-growing species such as maerl, horse mussels, cold-water coral reefs and *Serpula vermicularis* reefs. Slow or very slow recovering species within habitats that determine the recovery potential include long-lived slow growing Axinellid sponges, sea pens and anthozoans. Conversely, habitats that recover more rapidly are mobile sands, typical of high energy environments or other disturbed environments that are characterised by small, fast-growing, short-lived species. Mixed and muddy sediments, more stable sands and gravels and biogenic habitats characterised by faster growing species (seagrasses and kelps) are generally considered to have medium recovery potential.

Objective 3: Assisted recovery options

Habitat restoration encompasses a broad range of activities, emphasizing very different issues, goals, and approaches depending on the operational definition of 'restoration'. Building in mitigation measures to project designs is a precursor step to assist recovery by reducing the level of impact. Steps to assist recovery of natural habitats through mitigation management of pressures and coastal restoration measures have been widely implemented in the UK.

Passive recovery (removal of pressures) is required to support natural and assisted recovery. This is more difficult for pressures which are long-lived, including contamination, habitat change and the introduction of invasive non-native species. Pressures that have caused changes in habitat conditions (for example substratum change from sediment disturbance due to winnowing of fine sediments from fishing or removal on coarse fraction through aggregate extraction are also likely to lead to long-term effects in stable areas with low levels of hydrodynamic action or sediment budgets to support recovery.

Options to assist recovery were reviewed for habitats that are likely to recover more slowly, (recovery is assessed as Medium, Low or Very Low, based on the MarLIN categories). Approaches can be divided into eco-engineering options that improve physico-chemical factors and processes (including sediments, water quality and quantity) and those that engineer the ecology, by replanting or restocking species.

Eco-engineering approaches to restore the physical substratum include the use of natural rock blocks and boulders and the use of sediment capping, gravel and shell seeding can restore sedimentary habitats to previous conditions. Artificial reefs have been widely used in the marine environment, this approach may restore species assemblages but the physical habitat would differ from a natural rock habitat and this approach was not considered to restore natural rock habitats.

Assisted recovery options to restore species populations by restocking or transplanting have focussed on species that are described as ecological engineers that create biogenic habitats, these include seagrasses, kelps and bivalves (oysters and mussels).

There is some overlap between eco-engineering approaches and ecological approaches and both may be required to assist recovery. For example, sediment capping with clean sands and transplantation of seagrass have been proposed to restore organically enriched estuarine areas (Oncken and others, 2022).

Assisted recovery options may still require time for population recovery to a typical age structure and therefore, the assisted recovery timescales are generally the same as the natural recovery potential.

For most of the assessed EUNIS Level 4 habitats, no approaches were found to assist recovery of habitats and species assemblages. This agrees with previous work by Geist and Hawkins (2016) that recovery options in open marine systems are largely limited to managing human impacts in order to support natural recovery.

Objective 4: Assisted recovery costs, benefits, risks, challenges and uncertainties

For each approach, information on feasibility is provided in the appendices. Most approaches to assist recovery of marine species can still be regarded as under development in terms of application to English subtidal marine habitats. Application of most approaches is largely experimental and small-scale although seagrass and oyster restoration projects are beginning to be implemented in larger areas. As knowledge and experience of overcoming limiting factors and experience increases, assisted recovery feasibility is likely to improve.

A key challenge for assisted recovery approaches is the establishment of eco-engineering species. Where these are established they provide positive feedbacks that encourage recruitment and maintain populations. Biogenic habitat forming species such as kelp, seagrass and bivalves, stabilize and trap sediments and dampen wave energy facilitating retention of larvae and juveniles. Where these are lost, translocated individuals are typically either too small, too sparse or too unstable to modify their environment and establish such self-facilitating feedbacks.

Assisted recovery project typically are complex, resource intensive and have high costs. Cost-effective approaches that can be used over wider areas are being trialled, such as green gravel for kelps and seed bags for seagrass. As population size increases, connectivity between populations should support greater resilience. Studies found that while assisted recovery for sands and gravels which produce low levels of ecosystems services are not cost-effective, assisted recovery that provides high value services such as coastal protection or for habitats that provide high levels of ecosystem services or goods and benefits are likely to have more favourable cost-benefit ratios.

A number of risks are associated with assisted recovery. Approaches may impact donor populations (where stock or transplants are obtained, for example seagrass) and impact habitats and species within the footprint of the recovery project. Biosecurity risks around introduction or spread of pathogens and invasive non-native species are important for a range of projects that involve the movement of stock (transplantation and translocation) or infrastructure such as artificial beds. The creation of permanent or temporary infrastructure at sea will affect activities and other users and may impact safety.

Objective 5: Decision support framework

Many of the decisions around restoration projects and the specific actions to be undertaken are site-specific and cannot be accounted for in a generic framework. Nevertheless, we developed a five step framework to support advice and management options for Natural England that could support decision making workflows. The outputs of this assessment are provided in the Objective 5 Excel spreadsheet. Habitats within the UK Marine Habitat Classification that occur in inshore and offshore regions of England were assessed for natural recovery potential, relevant assisted recovery options and the feasibility, costs and benefits scored.

Given the low feasibility (see below), high costs and resources required for assisted recovery options, it is likely these would only be considered where pressures have been removed but populations are unlikely to recover naturally due to loss of connectivity and changes in habitat conditions (for example, negative feedbacks).

Limitations and uncertainties

Feasibility of approaches

Although the body of evidence and projects to support understanding is rapidly growing, feasible approaches for most habitats and species are not available or well-established. Application of assisted recovery over larger areas to improve ecology has not been carried out in the UK previously and most approaches should be considered as largely experimental and subject only to small scale trials. For all approaches, methodology and site selection are key to the feasibility of assisted recovery, this aspect may be resource intensive requiring physical investigations, modelling and monitoring of physico-chemical parameters, site ecology and the presence of risks such as invasive species. Any significant ecological, logistical, accessibility, legislative or financial barriers to the approach should be identified.

A key limitation for feasibility is the spatial scale of approaches. Recovery on small scales does not overcome negative feedbacks or support connectivity and the maintenance of self-sustaining, resilient populations but the costs and methodologies currently mean large-scale restoration is unfeasible for most habitats that have historically declined (such as oyster and seagrass beds).

Costs

Estimating costs is difficult, given that few authors have reported cost data to date. Of the costing exercises which have been carried out, the breakdown of costs has been inconsistently reported (Bayraktarov and others, 2016), limiting the scope for decision making on whether, what, how, where, and how much to restore. Costings are typically incomplete as they rarely include the following:

- investigation of impacts and managing or removing pressures to support recovery,
- research necessary to underpin decisions such as choice of appropriate techniques,
- selection of donor and restoration sites, and restoration landscape strategies (e.g. patch size)
- obtaining necessary licences, permits and liaising with stakeholders
- costs of non-consumable laboratory equipment (e.g. stereomicroscopes or diving materials) needed for these restoration projects are not included in estimates as pre-existing facilities are usually utilised
- costs associated with monitoring and maintenance

There is, therefore, a great deal of uncertainty in assessing costs and portioning these between capital, operational and maintenance costs. A number of assumptions were made by the project to address evidence gaps. For large scale, complex projects with multiple partners, the headline reported funding may not reflect all costs and value. For example, partners may make in-kind contributions and provide additional stakeholder work, scientific supervision, use of technicians and PhD students to conduct projects.

Conclusions

Assisted recovery approaches have clear value for restoring biogenic habitats that have undergone historic declines and which have not recovered naturally and that on their recovered state provide high levels of ecosystem services and goods and benefits. Significant barriers remain to assisting recovery including costs, feasibility, the complexity of projects and the level of resources required. However, approaches are being developed such as the hessian bag planting method for seagrass seeds and the green gravel approach for kelps, that are lower in cost and scalable to larger areas.

Marine assisted recovery options that are focussed on species are expensive and labour intensive. For most marine species, including larger burrowing infauna (holothurians, large polychaetes) and epifaunal species such as seapens and faunal communities of rock, these barriers, coupled with low levels of economic return mean that no options have been developed to assist recovery. It is therefore likely that for most habitats, the removal of pressures to support the recovery of degraded habitats and the management and conservation of remaining habitats will be prioritised over assisted recovery.

List of tables

List of figures

Appendix 1 Literature search terms

Search Term	Number of hits	Google Scholar (GS) or Google
Artificial rock reef UK	25,800	GS
Live maerl translocation	4,560	Google
Maerl restoration	35,200	Google
Mussel aquaculture OR relaying or seeding OR costs	3,990,000	Google
Sediment capping OR sealing OR disposal	19,000,000	Google
Sediment capping OR sealing OR disposal	491,000	GS
Maerl translocation	7,030	GS
Marine mud recovery	27,900	GS
Marine mud restoration	60,200	GS
Marine sediment capping	50,300	GS
marine subtidal assisted recovery	12,200	GS
Mytilus edulis aquaculture costs	16,600	GS
Cost benefit marine restoration	169,000	GS
Reef restoration UK	3,740,000	Google
Restor* marine chalk	340	GS
Restor* marine clay	2,540	GS
Restor* OR Recover Antedon	3	GS

Search Term	Number of hits	Google Scholar (GS) or Google
Restor* OR Recover Brachiopod	46	GS
Restor* OR Recover Neocrania OR Protanthea	157	GS
Restor* OR Recover sponge	4,260	GS
Restor* OR Recover OR Rehabilitate OR Enhance marine sand	4,280	GS
Restor* OR Recover OR Rehabilitate OR Enhance Axinellid sponge	20	GS
Restor* OR Recover OR Rehabilitate OR Enhance Musculus discors	11,400	GS
Restor* OR Recover OR Rehabilitate OR Enhance subtidal mud	3,340	GS
Restor* OR recover* OR enhance* Lophelia pertusa	513	GS
Restor* OR recover* OR enhance* Brissopsis lyrifera	63	GS
Restor* OR recover* OR enhance* sabellaria spinulosa	237	GS
Restor* OR recover* OR enhance* serpula vermicularis	151	GS
Restor* OR Recover* OR Rehabilitat* OR Enhance Branchiostoma lanceolatum	260	GS
Restor* OR Recover* OR Rehabilitat* OR Enhance Neopentadactyla mixta	23	GS
Restor* OR Recover* OR Rehabilitat* OR Enhance Ocnus planci		

Search Term	Number of hits	Google Scholar (GS) or Google
Restor* OR Recover* OR Rehabilitat* OR Enhance subtidal marine gravel	3,540	GS
Restor* OR Recover* OR Rehabilitat* OR Enhance subtidal mixed sediment	8,100	GS
Restor* OR Recover* OR Rehabilitate OR Enhance Arenicola		GS
Restor* OR Recover* OR Rehabilitate OR Enhance Ascophyllum	65	GS
Restor* OR Recover* OR Rehabilitate OR Enhance Cerastoderma edule	1,050	GS
Restor* OR Recover* OR Rehabilitate OR Enhance hiatella arctica	149	GS
Restor* OR Recover* OR Rehabilitate OR Enhance leptopsammia pruvoti	74	GS
Restor* OR Recover* OR Rehabilitate OR Enhance marine clay	2,527	GS
sand gravel aggregate site restoration	21,800	GS
Subtidal beneficial use dredge	48,500	Google

Notes. *=wild card character, e.g., Restor* = restoration, restoring, restorative etc.

Appendix 2 Recovery potential and relevant restoration approaches

Assisted recovery options identified for EUNIS Level 4 habitats with medium (full recovery within 2-10 years) (based on constituent Level 5 and/or 6 records) recovery to abrasion.

Table 11. Assisted recovery options identified for EUNIS Level 4 habitats with medium (full recovery within 2-10 years).

Medium Recovery	Recovery options
A4.22 *Sabellaria* reefs on circalittoral rock	*Rock habitats can be created using placement of blocks or similar. No approaches relevant to Sabellaria spinulosa*- were identified
A4.24 Mussel beds on circalittoral rock	Translocation of *Mytilus edulis*
A4.25 Circalittoral faunal communities in variable salinity	Rock habitats can be created using placement of blocks or similar. No approaches relevant to species assemblage
A4.31 Brachiopod and ascidian communities on circalittoral rock	Rock habitats can be created using placement of blocks or similar. No approaches relevant to species assemblage.
A5.27 Deep circalittoral sand	Landscaping techniques to create sandbars mimicking natural sand waves, have been used following sand extraction.
A5.35 Circalittoral sandy mud	Sediment placement or capping (contaminants)

Assisted recovery options identified for EUNIS Level 4 habitats with medium- high recovery ((full recovery within 2-10 years) to abrasion (based on constituent Level 5 and/or 6 records).

Table 12 Assisted recovery options identified for EUNIS Level 4 habitats with medium- high recovery.

Medium-High Recovery	Recovery options
A3. 11 Kelp with cushion fauna and/or foliose red seaweeds	Rock habitats can be created using placement of blocks or similar. Kelp and seaweed restoration approaches.
A3.12 Sediment-affected or disturbed kelp and seaweed communities	Rock habitats can be created using placement of blocks or similar. Kelp and seaweed restoration approaches.
A3.22 Kelp and seaweed communities in tide-swept sheltered conditions	Rock habitats can be created using placement of blocks or similar. Kelp and seaweed restoration approaches. A3.223 Boulder restoration.
A3.31 Silted kelp on low energy infralittoral rock with full salinity	Rock habitats can be created using placement of blocks or similar. Kelp and seaweed restoration approaches.
A3.36 Faunal communities on variable or reduced salinity infralittoral rock	Rock habitats can be created using placement of blocks or similar. Blue mussel, Mytilus edulis can be translocated.
A4.21 Echinoderms and crustose communities on circalittoral rock	Rock habitats can be created using placement of blocks or similar, artificial reefs may be engineered to enhance species assemblage. No directly approaches relevant to species assemblage.
A5.24 Infralittoral muddy sand	Sediment placement or capping (contaminants). A5.241: Experimental rearing of *Ensis sliqua* in hatcheries (da Costa, 2010). No methods *Echinocardium cordatum*.
A5.26 Circalittoral muddy sand	Sediment placement or capping (contaminants)
A5.33 Infralittoral sandy mud	Sediment placement or capping (contaminants)
A5.34 Infralittoral fine mud	Sediment placement or capping (contaminants)

Medium-High Recovery	Recovery options
A5.37 Deep circalittoral mud	Sediment placement or capping (contaminants)
A5.52 Kelp and seaweed communities on sublittoral sediment	Kelp and seaweed restoration approaches.
A5.53 Sublittoral seagrass beds	Seagrass restoration: translocation, seeding

Assisted recovery options identified for EUNIS Level 4 habitats with low (full recovery within 10-25 years) based on constituent Level 5 and/or 6 records.

Table 13. Assisted recovery options identified for EUNIS Level 4 habitats with low recovery

Low Recovery	Recovery options
A4.71 Communities of circalittoral caves and overhangs	No techniques identified.

Assisted recovery options identified for EUNIS Level 4 habitats with low (full recovery within 10-25 years) - very low (Negligible or prolonged recovery possible; at least 25 years to recover structure and function) based on constituent Level 5 and/or 6 records.

Table 14. Assisted recovery options identified for EUNIS Level 4 habitats with low or very low recovery.

Low -Very low Recovery	Recovery options
A4.12 Sponge communities on deep circalittoral rock	Rock habitats can be created using placement of blocks or similar, artificial reefs may be engineered to enhance species assemblage. No approaches relevant to Axinellid sponges
A5.51 Maerl beds	Translocation (note attempted for dead maerl but failed for live)
A5.63 Circalittoral coral reefs	Translocation of coral pieces
A5.71 Seeps and vents in sublittoral sediments	No approaches identified

Assisted recovery options identified for EUNIS Level 4 habitats with very low (Negligible or prolonged recovery possible; at least 25 years to recover structure and function) based on constituent Level 5 and/or 6 records.

Table 15. Assisted recovery options identified for EUNIS Level 4 habitats with very low recovery

Low or very low Recovery	Recovery options
A3.21 Kelp and red seaweeds (moderate energy infralittoral rock). **A3.2113- *Laminaria digitata* and piddocks on sublittoral fringe soft rock**	No restoration approaches for soft rock habitats identified. Kelp and seaweed restoration approaches would support restoration of that part of the species assemblage.
A3.34 Submerged fucoids, green or red seaweeds (low salinity infralittoral rock) **A3.342- *Ascophyllum nodosum* and epiphytic sponges and ascidians on variable salinity infralittoral rock**	Rock habitats can be created using placement of blocks or similar. Seaweed restoration approaches. *Ascophyllum nodosum* have been reported from artificial rock pools created in hard infrastructure.
A4.13 Mixed faunal turf communities on circalittoral rock **A4.1311- *Eunicella verrucosa* and *Pentapora foliacea* on wave-exposed circalittoral rock**	Rock habitats can be created using placement of blocks or similar.
A4.23 Communities on soft circalittoral rock **A4.231 Piddocks with a sparse associated fauna in sublittoral very soft chalk or clay**	No approaches identified to restore habitat or species assemblages.
A5.36 Circalittoral fine mud **A5.361 Seapens and burrowing megafauna in circalittoral fine mud**	No approaches identified to restore specific habitat and species (sediment capping and placement would be highly damaging)
A5.43 Infralittoral mixed sediments **A5.434 *Limaria hians* beds in tide-swept sublittoral muddy mixed sediment** **A5.435 *Ostrea edulis* beds on shallow sublittoral muddy mixed sediment**	Ostrea edulis restoration: improvements to brood stock, nurseries, translocation, habitat enhancement (cultch)

Low or very low Recovery	Recovery options
A5.44 Circalittoral mixed sediments **A5.442 Sparse *Modiolus modiolus*, dense *Cerianthus lloydii* and burrowing holothurians on sheltered circalittoral stones and mixed sediment**	No approaches identified to restore habitat.
A5.61 Sublittoral polychaete worm reefs on sediment **A5.613 *Serpula vermicularis* reefs on very sheltered circalittoral muddy sand**	No approaches identified to restore biogenic reef
A5.62 Sublittoral mussel beds on sediment	*Modiolus*: habitat improvements, cultch, translocation, *Mytilus edulis*, translocation
A6.61 Communities of deep-sea corals	Translocation possible but experimental technique

Appendix 3 Sand and gravel assisted recovery

Aggregate dredging of marine sands and gravels results in immediate removal of fauna and sediments and alters the seabed through creation of dredge furrows and pits. Changes in seabed composition result from the exposure of different underlying sediments or sediment screening, which returns unwanted sediment fractions, usually sands, to the seabed. The period of time such features persist will depend on hydro-dynamics, sediment particle size and the intensity of the activity (Foden and others, 2010). This section discusses a number of techniques used to restore the seabed following aggregate removal.

Dredging

Dredging unwanted material from the seabed using a conventional Trailer Suction Hopper Dredger. The dredged material can be used commercially as infill for dredge depressions, discharged at a disposal site or used for beneficial purposes such as beach recharge (Cooper and others, 2013).

Shell seeding

Aggregate dredging removes shell deposits that have accumulated as a surface veneer. This shell has taken thousands of years to accumulate and provides settlement surfaces for the larvae of benthic species. A study by Collins and Mallinson (2007) into the use of shell to speed the recovery of aggregate-dredged seabed assessed the use of both scallop shell and crushed whelk shell compared to 50mm stone and 20mm stone in an area of artificial reef within Poole Bay and an area to the east of the Isle of Wight on extracted aggregate seabed.

Gravel seeding

Cooper and others (2007, 2013) investigated the effectiveness, practicality and costs of gravel seeding as a potential method to restore the seabed at marine aggregate extractions sites. Gravel seeding was proposed to restore the composition of seabed sediments in areas characterized by an overburden of sands resulting from sediment screening.

Bed levelling

Bed levelling using a dredge plough can level high spots which remain after dredging. Whilst this approach is very effective in the context of maintenance dredging, where water depths are relatively shallow and sediments are comparatively soft.

EUNIS Habitats recovery option is applicable to:

Broadly, these approaches may be suitable for restoring the broadscale habitats A5.1 Sublittoral coarse sediment and A5.2 Sublittoral sands.

Natural recovery summary for applicable habitats

The reported times for the recovery of biological resources after aggregate dredging vary from months to decades. For sands and gravels in high energy, naturally disturbed environments, physical and biological recovery is rapid (2-4 years) because dredge tracks are quickly eroded and faunal communities are made up of many small bodied, rapidly maturing opportunistic species that are already adapted to high levels of disturbance and rapidly recolonize disturbed areas. For example, in areas such as the Bristol Channel, where dredging has been carried out in mobile sandy habitats, the physical impacts of dredging have been observed to disappear within a few tidal cycles (Newell and others, 1998). In deeper, stable areas, recovery of long-lived species may take up to 15 years (Bellew and Drabble, 2004).

Where sediment composition has been altered, the original species assemblage may not be able to recover. Boyd and others (2004) compared dredging at several east coast sites in the UK, and found that the intensity of the dredging activity influenced the rate of recovery for the physical characteristics of the substratum, as well as its benthic community. Scarring/ tracks were still evident 3-10 years after dredging (Boyd and others, 2004). Foden and others (2009) found that recovery times in coarse sand and gravel habitats were related to tidal currents with faster recovery times being associated with stronger currents (although this relationship was not necessarily linear). Physical recovery times ranged from 5 to 20 years although mean biological recovery times of 5-12 years were reported.

Overview of approach advantages when compared to natural recovery

It is assumed that the timescale for recovery of the biological assemblages following assisted recovery of sediments would be equivalent to natural recovery potential from extraction which results in loss of the species assemblage.

Shell seeding promoted rapid recovery of species richness. In only 7 months, the shell species richness had achieved 70% of that on mature recovering dredged aggregate seabed (Collins and Mallinson, 2007).

Overview of approach disadvantages compared to natural recovery

Assisted recovery options require capital and operation inputs. Options that use gravel must have sourced this with impacts on other gravel habitats in the borrow site. Introduction of shell material should be subject to biosecurity measures to prevent introduction of invasive non-natives.

Sand and gravel assisted recovery benefits

As part of cost-benefit analysis, Cooper and others (2013) valued the ecosystem service benefits at an impacted and an unimpacted reference site. These values demonstrate the ecosystem service benefits associated with restoration. The paper should be referred to for methodological data. Other ecosystem services identified but not valued include the formation of species habitat for fish and shellfish,

Formation of species habitat: Shell seeding. The Poole Bay study found that a similar number of species colonized the scallop shell and larger stone, although the scallop shell was more densely settled and provided habitat niches for mobile fauna. In the Isle of Wight study area, scallop shells provided ideal habitat for prawns and squat lobsters. Small fish (cling fish and gobies) and porcelain crabs also found to shelter in shells lying flat on the seabed. Scallop shells proved to be very successful at promoting fast colonization. Of the 102 macrofaunal species identified, 14 species were found only on the newly deployed shell and were not found on the aggregate, even after over 5 years.

Carbon sequestration. Cooper and others (2013) assessed that the value of carbon sequestration ranges from £58.50 to £148.29 per km^2 to £6.79 to £46.30 per km^2. The value at the non-impacted (reference) site ranged between £81.13 and £186.69 per km^2.

Education relating to the marine environment. Cooper and others (2013) estimated that £187,500 of funded research has taken place (between 2000 and 2012) specifically referring to the impact of dredging at Area 222 and such studies have included several research papers.

Sand and gravel assisted recovery: economic costs

Cooper and others (2013) undertook an economic analysis of site restoration using a mixture of techniques. Area 222 is an historic aggregate extraction area occupying 0.3383 km^2. The site is located in the outer Thames Estuary, in water depths of 27–35 m. The estimated predicted total cost of restoring the site was £712 k–>£1 million, depending on whether natural recovery occurred following the removal of a sand wave feature near the experimental site. On balance, the analysis indicated that the restoration of the sea bed was not justified due to high implementation costs. The authors stress that this was a site-specific decision that may not be applicable to all situations.

Licensing costs. Any restoration actions placing material on the seabed require a marine licence.

Survey costs: include a minimum of two surveys plus any normal post-dredge survey. The first survey assesses the significance of impacts and the extent of recovery, provides data to allow development of a detailed restoration plan, and forms a baseline against which the success of restoration may be judged. This survey would include a full coverage acoustic survey of the site, together with ground truthing using 0.1 m^2 Hamon grab, camera and 2 m beam trawl samples. A minimum of 10 sample grab replicates would be required from within each impact zone and reference site (Cooper and others, 2011). Following restoration works, a ‘post-restoration’ survey would establish whether the work had been successful. As the restoration aims to address physical changes, this survey would not include a biological component. The cost of survey work includes vessel time, staff time, sample processing, and data analysis/reporting. The costs for surveys are £54,834 for the baseline survey, £20,600 for the post-restoration survey, and £14,600 for the post-restoration survey.

The overall costs of restoring Area 222 were assessed according to restoration works (per zone), licensing, carbon footprint and survey work. The total site costs are presented in simplified form in the Table 16 below. Total cost was converted to cost per km^2 based on the site (0.34km^2) and then to hectares to allow comparison with other restoration activities. These costs should be considered cautiously bearing in mind that it is not clear that costs such as licensing are directly saleable to different areas.

No economic costs were reported for shell seeding. The seeding approach was considered to have a similar cost to gravel seeding based on operational similarities. The cost of shell material per hectare could be approximately £154,000/ha (see Appendix 8).

Table 16. Estimated costs of restoration works taken from Cooper and others (2013).

Action	Cost	Cost/km2	Cost/ha (2022 prices)
Dredging	193,763	561,912	5,619
Gravel seeding	653,840	1,923,058	19,230
Bed levelling	209,000	614,705	6,147
Licensing	39,700	116,764	1,167
Survey-baseline	54,834	161,276	1,612
Survey-post restoration	20,600	60,588	606
Summary cost dredging (incl. licensing and surveys: baseline and post-restoration)	308,897	950,540	9,004 (*£10,565)
Summary cost gravel seeding (incl. licensing, surveys for baseline and post-restoration)	768,974	2,261,686	22,615 (*£26,537)
Summary cost bed levelling (incl. licensing, surveys for baseline and post-restoration)	324,134	953,333	9,532 (*£11,185)
*Converted 2022 cost based on the inflation rate in the United Kingdom between 2013 and today as 17.34% (www.inflation tool.com)			

Sand and gravel assisted recovery: feasibility

Feasibility: The assisted approaches involve deploying materials or using dredge ploughs. These are technically feasible and involve single operations which do not require complex infrastructure or ongoing maintenance.

Feasibility: Bed levelling. Its effectiveness in the typically deeper water and coarser sediments of aggregate extraction areas is largely unproven (Cooper and others, 2013). Preliminary results from an extraction area in French waters did not show any obvious physical effect resulting from levelling using a 5 m plough (cited from Cooper and others 2013).

Feasibility: shell seeding. The UK shell processing industry produces sufficient quantities of shell annually to make the use of shell a realistic option to enhance the seabed after the cessation of marine aggregate extraction.

Gravel seeding. Initial findings from experiments at Area 408 (offshore Humber) suggest that gravel seeding operations were effective in restoring the sediment composition of the study area and increasing the proportion of gravel exposed at the seabed surface. This increase in gravel led to the establishment of a faunal community more similar to that of local gravel dominated reference sites. However, almost two years after the addition of gravel the sediments became sandier again (Hill and others, 2011).

Sand and gravel assisted recovery: risks, challenges and uncertainties

Risk: Invasive non-native species. Shell seeding. In the Poole Bay study area large numbers of juvenile invasive non-native slipper limpets, *Crepidula fornicata*, settled on the undersides of the scallop shells. Introduction of shell may also result in biosecurity risks (see Appendix 6).

Challenges: shell seeding: The selection of type and size of material is important. In the Poole Bay study the crushed whelk shell and smaller stone were colonized by half the number of species of the larger stone and scallop shell. In the Isle of Wight study the crushed whelk shell was also not successfully colonized. The unsuccessful nature of whelk shell colonization was due to its high mobility.

Sand and gravel assisted recovery: references

BELLEW, S., DRABBLE, R., 2004. Marine Aggregate Site Restoration and Enhancement Strategic Policy Overview. Report to the British Marine Aggregates Producers Association, the Crown Estate and English Nature (Ref: 04/J/01/06/0548/0437).

BOYD, S.E., COOPER, K.M., LIMPENNY, D.S., KILBRIDE, R., REES, H.L., DEARNALEY, M.P., STEVENSON, J., MEADOWS, W.J. AND MORRIS, C.D., 2004. Assessment of the re-habilitation of the seabed following marine aggregate dredging. Science series technical report. Centre for Environment, Fisheries and Aquaculture Science, 121, p.154.

COLLINS, K. AND MALLINSON, J. 2007. The use of shell to speed recovery of dredged aggregate seabed. School of Ocean and Earth Science, University of Southampton, National Oceanography Centre.

COOPER, K., BURDON, D., ATKINS, J.P., WEISS, L., SOMERFIELD, P.J., ELLIOTT, M., TURNER, K., WARE, S. AND VIVIAN, C., 2011. Seabed Restoration following marine aggregate dredging: Do the benefits justify the costs? MEPF-MALSF Project 09-P115. MALSF Lowestoft. 111pp. ISBN 978 0 907545 64 4. Contract Report, pp.1-111.

COOPER, K.M., WARE, S., VANSTAEN, K., AND BOYD, S.L. 2007 Gravel-seeding – A suitable technique for restoration of the seabed following marine aggregate dredging? In Marine Aggregate Dredging: Helping to determine good practice. Marine Aggregate Levy Sustainability Fund (ALSF). Conference Proceedings: September 2006.

COOPER, K., BURDON, D., ATKINS, J.P., WEISS, L., SOMERFIELD, P., ELLIOTT, M., TURNER, K., WARE, S. AND VIVIAN, C., 2013. Can the benefits of physical seabed restoration justify the costs? An assessment of a disused aggregate extraction site off the Thames Estuary, UK. Marine pollution bulletin, 75(1-2), pp.33-45.

FODEN, J., ROGERS, S.I. AND JONES, A.P., 2010. Recovery of UK seabed habitats from benthic fishing and aggregate extraction—towards a cumulative impact assessment. Marine Ecology Progress Series, 411, pp.259-270.

HILL, J. M., MARZIALETTI, S. AND PEARCE, B. 2011. Recovery of Seabed Resources Following Marine Aggregate Extraction. Marine ALSF Science Monograph Series No. 2. MEPF 10/P148. (Edited by R. C. Newell AND J. Measures). 44pp. ISBN: 978 0 907545 45 3.

NEWELL, R.C., SEIDERER, L.J. AND HITCHCOCK, D.R., 1998. The impact of dredging works in coastal waters: a review of the sensitivity to disturbance and subsequent recovery of biological resources on the sea bed. Oceanography and Marine Biology: an annual review, 36(1), pp.127-178.

Appendix 4. Sediment capping

Sediment capping has been used to restore contaminated sediments in areas with highly polluted surface sediments by sealing these through capping with a layer of clean sediment (Oncken and others, 2022; Schaanning and others, 2021; Eek and others, 2008). This is an alternate approach to dredging and removal of contaminated sediments which remove pollutants permanently and reduce sediment contaminant concentrations but are costly and can cause remobilization of contaminants that trapped in the sediments, rendering them more bioavailable (Zhang and others, 2016).

Sand-capping has previously been attempted in harbours to reduce dispersal of pollutants having been identified as a cost-effective technique for on-site remediation (Mohan and others, 2000). Techniques include both thick (50cm layers) and thin layer capping (5-10cm) with a range of sediment and capping materials (Näslund and others, 2012; materials like sand, silt, clay, and crushed rock debris and active materials (activated carbon, apatite, zeolite, organoclay), caps may be armoured with rock or artificial mattressing (Zhang and others, 2016).

Sediment capping has been used to cap organically enriched muds with a 10cm layer of clean sands at two locations (1.0 and 1.4 ha) in Odense Fjord, Denmark. To cap the sediments a movable, anchorable floating platform was used with an excavator placed on the platform allowing high precision deposition of sediments, while a barge provided sand for the capping action from a geological sand formation outside the estuary. The sand-cap stabilised the mud and the sand-mud interface persisted without mixing (after one year). The associated lower resuspension of fine particle improved light conditions in the overlying water by up to 9 and 22% at the two locations. Benthic fauna recruitment improved after sand-capping, leading to a local shift from low to high diversity of the benthic community and increased ecosystem functionality (Oncken and others, 2022).

EUNIS Habitats recovery option is applicable to

EUNIS sedimentary habitats.

Spatial scale and timescale to assisted recovery vs natural recovery

Natural recovery of contaminated sediments in basins takes ‘several’ years based on relatively low natural sediment accumulation rates (< about 2 mm/year), run-off from contaminated land areas, the time scales of water renewals, bioturbation and other physical disturbance (e.g., ship traffic remobilizing contaminated sediments), and supply of air-born contaminants (Alve and others, 2009).

Organic enrichment may persist for longer timescales than contaminants. The organic pools in enriched sediments will take decades to degrade naturally by microbial activity, with the easily resuspended organic matter increasing turbidity and preventing recovery of seagrass habitats and reducing benthic diversity (Oncken and others, 2022).

Natural recovery summary for applicable habitats

Recovery of sediments and associated assemblages will depend on site specific sediment transport, remediation of contaminants and recovery of the biological assemblage. No specific recovery rates have been provided here due to the wide range of EUNIS Level 4 habitats and constituent Level 5 and 6 child biotopes that could be relevant.

Overview of approach advantages when compared to natural recovery

This approach would be used where natural recovery of sediments is not possible and has the advantage, where necessary, of sealing contaminants or reducing fine particle resuspension. Natural recovery approaches would require bioremediation, burial and storage or transport of contaminants.

Overview of approach disadvantages compared to natural recovery

This approach would have impacts on donor sites from which sediments were removed. Impacts on species assemblages and connected habitats through changes in sediment composition following removal, sediment transport to connected habitats should be carefully considered. Altering sediments will change habitat suitability for benthic organisms. Sand capping may increase favourability for seagrass, for example, but reduce suitability for other species (Flindt and others, 2007).

Economic costs: Approach costs

Costs for sediment capping include labour, shipping, dredging equipment and fuel. Costs are offset by beneficial re-use of dredged material and may reduce operation costs for dredging where sediment transport distance is reduced (Flindt and others, 2022). Applying caps to contaminated sediment is typically less expensive than dredging, with costs depending primarily on cap design (Perelo, 2010).

Feasibility

Sand capping can beneficially re-use sands that are dredged to maintain navigation channels (Flindt and others, 2007) and can therefore be a cost-effective approach.

Risks, challenges and uncertainties

Site feasibility is key. Local hydyodynamics and sediment movement should be considered in terms of dispersal of caps or siltation. Stability of the sediments in a Danish study, varied by site. In a sheltered area the sand cap persisted for a year (duration of monitoring) but in a more dynamic site the sand cap was covered by a 3-5cm mud deposit (Oncken and others, 2022).

Sediment capping: References

ALVE, E., LEPLAND, A., MAGNUSSON, J. AND BACKER-OWE, K. 2009. Monitoring strategies for re-establishment of ecological reference conditions: possibilities and limitations. Marine Pollution Bulletin, 59(8-12), pp.297-310.

EEK, E., CORNELISSEN, G., KIBSGAARD, A. AND BREEDVELD, G.D., 2008. Diffusion of PAH and PCB from contaminated sediments with and without mineral capping; measurement and modelling. Chemosphere, 71(9), pp.1629-1638.

FLINDT, M.R., ONCKEN, N.S., KUUSEMAE, K., LANGE, T., AASKOVEN, N., WINTER, S., SOUSA, A.I., RASMUSSEN, E.K., CANAL-VERGES, P., CONNOLLY, R.M. AND KRISTENSEN, E. 2022. Sand-capping stabilizes muddy sediment and improves benthic light conditions in eutrophic estuaries: Laboratory verification and the potential for recovery of eelgrass (Zostera marina). Journal of Sea Research, p.102177.

MOHAN, R.K., BROWN, M.P. AND BARNES, C.R., 2000. Design criteria and theoretical basis for capping contaminated marine sediments. Applied ocean research, 22(2), pp.85-93.

NÄSLUND, J.S.G.S., SAMUELSSON, G.S., GUNNARSSON, J.S., NASCIMENTO, F.J., NILSSON, H.C., CORNELISSEN, G. AND SCHAANNING, M.T. 2012. Ecosystem effects of materials proposed for thin-layer capping of contaminated sediments. Marine Ecology Progress Series, 449, 27-39.

ONCKEN, N.S., LANGE, T., KRISTENSEN, E., QUINTANA, C.O., STEINFURTH, R.C. AND FLINDT, M.R., 2022. Sand-capping–A large-scale approach to restore organic-enriched estuarine sediments. Marine environmental research, 173, p.105534.

PERELO, L.W., 2010. In situ and bioremediation of organic pollutants in aquatic sediments. Journal of hazardous materials, 177(1-3), pp.81-89.

SCHAANNING, M.T., BEYLICH, B., GUNNARSSON, J.S. AND EEK, E., 2021. Long-term effects of thin layer capping in the Grenland fjords, Norway: Reduced uptake of dioxins in passive samplers and sediment-dwelling organisms. Chemosphere, 264, p.128544.

ZHANG, C., ZHU, M.Y., ZENG, G.M., YU, Z.G., CUI, F., YANG, Z.Z. AND SHEN, L.Q., 2016. Active capping technology: a new environmental remediation of contaminated sediment. Environmental Science and Pollution Research, 23(5), pp.4370-4386.

Appendix 5. Boulder field restoration

The first attempt to restore a large-scale marine boulder reef in Europe was initiated in the Kattegat in 2008 and a number of follow-on studies were undertaken (Stenberg and others, 2015, Støttrup and others, 2017; Støttrup and others, 2014). The restored reef covered approximately 27,600 m^2 seafloor and included 100,712 tons of boulders of varying size added at depths ranging between 4 and 11 m. Measured at the seabed surface, the mean depth of the study area decreased from 7.6 m before the deposition of the boulders to 6.6 m after deposition (Stenberg and others, 2015). Data on the physical structure of the restored boulder reef, collected in 2009, demonstrated that cavernous structures and shallow reef areas were restored. Moreover, data collected in 2012 confirmed the stability of the restored reef. The objective of the project was to restore ecological reef functions and services and achieve a favourable conservation status to fulfil international obligations for the reef habitat type (Mikkelsen and others, 2013, Stenberg and others, 2015, Støttrup and others, 2014). The project was initiated in 2007 and the boulders deployed in 2008. The restoration objectives were to: 1) stabilize the uppermost part of the study area, 2) restore crevices of different sizes, 3) restore a diverse topography with varying heights and inclinations and 4) restore former shallow reef structures to support functions and services of the habitat. Furthermore, to ensure sustainability and stakeholder support, the project aimed to increase both public awareness and scientific knowledge of protecting and restoring boulder reef habitats.

EUNIS Habitats that boulder field restoration is applicable to:

- A3.123 *Laminaria saccharina*, *Chorda filum* and dense red seaweeds on shallow unstable infralittoral boulders and cobbles;
- A3.2112 *Laminaria digitata* and under-boulder fauna on sublittoral fringe boulders;
- A3.223 Mixed kelp and red seaweeds on infralittoral boulders, cobbles and gravel in tidal rapids;
- A4.137 *Flustra foliacea* and *Haliclona oculata* with a rich faunal turf on tide-swept circalittoral mixed substrata, and
- A6.14 Boulders on the deep-sea bed.

Boulder restoration may also be used in conjunction with restoration of bivalve reef (Liversage, 2020).

Natural recovery summary for applicable habitats:

Recovery over large scales following the loss of boulders due to quarrying or deliberate removal will not occur naturally, with the exception of localised losses in areas where wave action or currents are able to mobilise boulders,.

Overview of approach advantages when compared to natural recovery

Natural recovery will not occur except in areas where water movement can mobilise boulders. Therefore, this approach is required to restore habitats.

Overview of approach disadvantages compared to natural recovery

The use of boulders that are not composed of local stone (or similar hard substratum) may alter the species assemblage due to the change in habitat. Tests with stone panels fixed to the sublittoral, mid-tide and high-tide levels of varying roughness found that *Ulva* species settled preferentially on smoother, fine grained, substratum (chalk, mottled sandstone) and *Porphyra purpurea* on rougher, granulated substratum (limestone, granite, basaltic larvae) (Luther, 1976). *Corallina officinalis* shows optimal settlement on finely rough artificial substrata (0.5 - 1 mm surface particle diameter). Although spores will settle and develop as crustose bases on smooth surfaces, fronds were only initiated on rough surfaces. Crustose coralline algae extend further to the undersides of natural, rounded boulders than experimental stone blocks likely due to availability of light (Liversage, 2016).

Boulder field restoration economic costs

The total budget for the above project in the Kattegat was € 4,808,398 (based on online LIFE funding website[2]), it is not clear how this was apportioned to activities associated with the project.

Capital costs

Capital costs included:

- feasibility studies including:
    - diver observations of seabed,
    - geotechnical investigations including drilled profiles and seismic surveys,
    - hydrodynamics;
    - wave simulations including wave height modelling,
    - sediment transport modelling, and
- Stakeholder mapping and engagement.

Operation costs

Operational costs included the purchase of boulders and their transport by barge and stakeholder engagement.

Stakeholder engagement costs: During the construction phase and until a new sea chart was published, six sea markers were deployed banning all shipping within the area. An information campaign targeted the yachting community, enforced with billboard information at two harbours on Læsø and in harbours in surrounding areas (Frederikshavn and Sæby), and letters warning about the depth changes were sent to yachting organizations in Denmark and Sweden.

Information on the project was disseminated through a pamphlet and several articles in local newspapers and two posters exhibited on Læsø. An article on the project was published in a tourist magazine and two programs were broadcasted on national television.

Boulder field assisted recovery benefits

Formation of species habitat and food: A study of 33 tagged cod from the boulder reef restoration project of Læsø Trindel in Kattegat, found that a larger fraction of the released fish

[2] https://webgate.ec.europa.eu/life/publicWebsite/index.cfm?fuseaction=search.dspPage&n_proj_id=3109

remained in the study area after restoration (94%) than before (53%). Moreover, throughout the study period, cod spent significantly more hours per day and prolonged their residence time in the study area (Kristensen and others, 2017). The study indicated that marine reefs subjected to boulder extraction could be restored and function as favourable cod habitats.

Natural Hazard regulation/ Prevention of coastal erosion: Boulders or cobbles may be ideal artificially-introduced ‘roughness elements’ that mimic the function of large shells by creating different area of turbulence, including less-turbulent areas that may support settlement of larvae (Liversage, 2020).

Boulder field assisted recovery feasibility

The mobility of boulders means these reefs can be restored or created with relative ease compared to other rocky marine habitats such as rock-platforms. Boulder-reef restoration studies have been undertaken globally and involve numerous rock-types and other artificial hard-substrata (Liversage, 2020). The approach is not effective for areas with mobile fine sediments that could smother boulders (Liversage, 2020).

In the Kattegat study, the geological and geotechnical surveys before the restoration, confirmed that the sea bed could support added boulders, and high resolution bathymetric surveys provided input for the design of the reef, particularly for numerical modelling of the hydrographic and sediment transport conditions. Numerical modelling was used to derive hydrographic design conditions for boulder placements and further, to ensure that the restored reef would not affect the sea bed morphology and hydrographic conditions at a local harbour and at a protected habitat, both situated in the vicinity of the restoration area. Data on the physical structure of the restored boulder reef, collected in 2009, demonstrated that cavernous structures and shallow reef areas were restored. Moreover, data collected in 2012 confirmed the stability of the restored reef.

Boulder field assisted recovery: risks, challenges and uncertainties

Challenges: baselines: Old bathymetry maps can provide guidance but may also be inaccurate or unavailable. Moreover, the sizes and shapes of removed boulders are usually unknown, and subsequent erosion may have changed the remaining seabed. Therefore, restored reefs typically constitute novel structures in the marine environment that are exposed to hydrodynamic forces from waves and currents. Therefore, reef restoration should follow the same procedures as most other marine construction works with regards to planning and deployment. Importantly, hydrodynamic forces have the potential to modify a reef or render it unstable, and appropriate precautions should cover reef designs developed with engineering tools including numerical modelling of local conditions. Furthermore, local geotechnical properties may limit the carrying capacity of the sea bed and may therefore constrain the design or the choice of foundation for the restored reef. The stability of the projected reef must be ensured to align with the main objectives of restoring lost ecosystem functions and services in the design phase of the restoration process,. Unstable reef structures will be prone to degradation over time, and possibly prevent the restored reef’s ability to meet the intended objectives (Støttrup and others, 2017).

Risks: changes in hydrodynamics. Large reef structures may change local current and wave patterns and potentially affect sediment transport and thereby seabed morphology. Possible impacts may extend to coastal areas (for example, beaches or harbours), similar to other coastal structures, depending on the geometrical properties of the reef, and the location of the reef (S Støttrup and others, 2014). The design phase must ensure that the restored reef will not impact other protected habitats in the vicinity of the restored reef. For example, it is crucial that a restored reef does not cause elevated sedimentation in neighbouring protected habitats.

Risks to recreational vessels: Despite the clear marking, several recreational sailing vessels collided with the new and shallower reef established in the Kattegat. To avoid further accidents, an information campaign targeting the yachting community was enforced with billboard information at two harbours on Læsø, in harbours in surrounding areas (Frederikshavn and Sæby) and letters warning about the depth changes were sent to yachting organisations in Denmark and Sweden.

Risks: **Non-native species.** Introduced structures, such as new reef, has the potential to establish inappropriate habitat corridors (for example, invasive species).

Boulder field assisted recovery: references

KRISTENSEN, L.D., STØTTRUP, J.G., SVENDSEN, J.C., STENBERG, C., HØJBJERG HANSEN, O.K. AND GRØNKJÆR, P., 2017. Behavioural changes of Atlantic cod (Gadus morhua) after marine boulder reef restoration: implications for coastal habitat management and Natura 2000 areas. Fisheries Management and Ecology, 24(5), pp.353-360.

LIVERSAGE, K., 2016. The influence of boulder shape on the spatial distribution of crustose coralline algae (Corallinales, Rhodophyta). Marine Ecology, 37(2), pp.459-462.

LIVERSAGE, K., 2020. An example of multi-habitat restoration: Conceptual assessment of benefits from merging shellfish-reef and boulder-reef restorations. Ecological Engineering, 143, p.105659.

LUTHER, G., 1976. Bewuchsuntersuchungen auf Natursteinsubstraten im Gezeitenbereich des Nordsylter Wattenmeeres: Algen. Helgoländer Wissenschaftliche Meeresuntersuchungen, 28 (3-4), 318-351.

MIKKELSEN, L., MOURITSEN, K. N., DAHL, K., TEILMANN, J., AND TOUGAARD, J. 2013. Re-established stony reef attracts harbour porpoises *Phocoena phocoena*. Marine Ecology-Progress Series, 481, 239–248

STENBERG, C., STØTTRUP, J., DAHL, K., LUNDSTEEN, S., GÖKE, C., AND ANDERSEN, O. N. 2015. Ecological benefits from restoring a marine cavernous boulder reef in Kattegat, Denmark. DTU Aqua report no. 289-2015. 49 pp

STØTTRUP, J. G., STENBERG, C., DAHL, K., KRISTENSEN, L. D., AND RICHARDSON, K. 2014. Restoration of a temperate reef: Effects on the fish community. Open Journal of Ecology, 4, 1045–1059

STØTTRUP, J.G., DAHL, K., NIEMANN, S., STENBERG, C., REKER, J., STAMPHØJ, E.M., GÖKE, C. AND SVENDSEN, J.C., 2017. Restoration of a boulder reef in temperate waters: Strategy, methodology and lessons learnt. Ecological Engineering, 102, pp.468-482.

Appendix 6. Rock reef assisted recovery

Artificial reefs have long been used in the marine environment and are created from a wide range of materials (Baine, 2001) for a wide range of purposes (Jensen, 2002). Artificial reefs created for aquaculture/sea-ranching, biomass increase, biodiversity enrichment, fisheries production, ecosystem management, prevention of coastal erosion, recreational activities (for example, scuba diving, ecotourism, fishing), and research are typically constructed to resemble natural reefs as much as possible, with the ultimate goal to produce similar effects (Glarou and others, 2020). Marine infrastructure such as break walls, pier pilings and offshore wind farms (OWFs) create artificial reefs (Morris and others, 2018). Scour protection typically, made of gravel, shielded by a rock armour layer is placed around OWF turbines and cables (Glarou and others 2020).

There is a considerable body of research (O'Shaughnessy and others 2020) that has explored ecological enhancement of marine structures, particularly, for example, rock armour defences and some vertical wall structures (piles, quay walls), although the actual application of research ideas has often been more limited. In the UK to date, ecological enhancement has tended to be undertaken at very local, small, scales, examples include:

- Incorporation of rock pools into quay wall designs (for example, Wightlink, 2018);
- Creation of artificial rockpools and textured surfaces to enhance colonization (for example, porous blocks) (for example, Firth and others, 2016; Naylor and others, 2011, 2017)
- Creation of artificial reefs (for example, Fabi and others, 2011);
- West Sussex, UK, Added pits (large and/or small) to seawall (Moschella and others, 2005)
- Shaldon, UK Added grooves, pits, and recessed crevices Firth and others (2014)
- Colwyn Bay, UK Added Bioblock unit (Firth and others, 2014)

As the focus of this project is on assisted recovery of habitats, the review has focussed on examples that are more relevant to rock habitats using either hard substratum including concrete and rock and scour protection in hard substratum habitats rather than artificial structures which host species assemblages that are distinct from natural rock reefs for example, the steel wreck HMS Scylla sunk to create an artificial diving reef (Hiscock and others 2010) and the geotextile/sand artificial surfing reef in Poole bay (Herbert and others 2017). However, such artificial surfaces may support biodiversity by providing habitat in areas of soft sediment for hard substratum species, for example, *Caryophyllia smithii* in the North sea (Coolen and others 2015).

A list of UK examples follows.

1) An artificial reef in Poole Bay reef made from cement stabilized pulverized fuel ash and flue gas desulphurization gypsum (Jensen and others 2000).
2) A quarry rock reef off the east coast of Scotland (Todd and others, 1992). The Torness reef was constructed from quarried rock derived from the construction of a nuclear power station. No studies on this reef could be found, despite searches.
3) Loch Linnhe (on the Scottish west coast, started in 2001). A large-scale reef complex being constructed on the west coast of Scotland and intended to research interactions between the reef structure and associated animals (with emphasis on lobsters) and plants, and the influences on the physical environment (Sayer and Wilding 2002; Wilding and Sayer, 2002a,b). The Loch Linnhe reef is constructed from blocks formed from cement stabilized quarry dust slurry, effectively recycling an inert waste material into an artificial reef. When completed, the reef will have 42 000 t of material deployed in 24 modules.

4) Salcombe (southwest England), where a natural rock reef was placed in 2000. This site was referenced by Challinor and Hall (2008) but detail was very limited and no further information could be found.

Other projects that are not focussed on restoration but which use similar techniques are offshore energy projects that use rock armouring for cable and scour protection. An example is Wave Hub, an 8 km^2 marine renewable energy infrastructure located off the north coast of Cornwall, south west UK. A 25 km subsea power cable, was protected by rock armouring, at a minimum burial depth of 30 cm, with concrete mattressing at 120 m intervals to provide additional stabilization as the substratum was not suitable for trenching. Overall, 80,000 tonnes of rock was deployed on the seabed (Sheehan and others 2020). Owing to the similarity in substratum between the cable rock armouring and surrounding habitat, the colonizing species on the cable were also present in the controls, despite significant differences in assemblage composition between treatments. Five years after deployment, the cable was supporting an epibenthic community that was becoming congruent with the surrounding ecosystem (Sheehan et al 2020).

EUNIS Habitats recovery option is applicable to:

Broadly, artificial rock reefs may be suitable for restoring infralittoral rock (EUNIS A.30 and circalittoral rock (EUNIS A.4 habitats). Its application to restore specific species assemblages (for example to restore hydroids but not bryozoans has not been explored).

Natural recovery summary for applicable habitats:

The recovery over large scales following the loss of rock reef will not occur naturally, with the exception of localised losses in areas where wave action or currents are able to remove sediments that have smothered reefs.

Overview of approach advantages when compared to natural recovery

Natural recovery will not restore rock reefs where these have been quarried.

Overview of approach disadvantages compared to natural recovery

Restoration of reef habitat would facilitate recovery when this is not possible through normal ecosystem processes.

Rock reef assisted recovery: economic costs

No information was found on costs.

Rock reef assisted recovery: benefits

Formation of species habitat and food. The Torness reef has been shown to influence local populations of cod *Gadus morhua* that probably use the reef as shelter rather than a source of food. The local lobster *Homarus gammarus* population may have been enhanced, while edible crab catch numbers do not appear to have been influenced. The presence of macroinvertebrates such as whelks, urchins, and starfish all reflect the habitat provided by the reef. The authors stressed the importance of an extended survey period in assessing reef influence on catches (Todd and others, 1992, cited from Jensen, 2002). The natural reef at Salcombe supports a range of species including anemones, sponges, ascidians, and cup corals (Challinor and Hall, 2008).

Cost benefit analysis summary:

No information on costs was found against which to assess benefits. Costs, particularly for broadscale application are likely to be high as the approach will require extensive project planning and operation costs: maintenance costs are considered to be none, as restoration would typically require one-off installation.

Rock reef assisted recovery: feasibility

The approach requires input of rock, transportation and deployment. This technique is likely to be only feasible at small scales in areas that are relatively shallow. The approach is considered to be feasible as artificial reefs and rock or artificial hard substratum armouring are widely used to protect cables.

Rock reef assisted recovery: risks, challenges and uncertainties

Reef restoration requires input of quarried materials for natural rock restoration. Quarrying will have impacts on donor sites. The physical and chemical stability of materials including natural rock, should be considered. Some natural rock may contain metals which can leach and be bioaccumulated by marine life (Challinor and Hall, 2008).

While reefs may be restored, no methods were identified that control the structure and function of the associated biological assemblage. While species composition may be predicted based on nearby rock habitats, the associated species that colonise and establish are likely to undergo changes over time (Sheehan and others 2020).

Rock reef assisted recovery: references

BAINE, M. 2001. Artificial reefs: a review of their design, application, management and performance. Ocean Coast. Manag. 44, 241–259. doi: 10.1016/S0964-5691(01)00048-5

CHALLINOR, S. AND HALL, H., 2008. Multi-functional artificial reefs scoping study. CIRIA C673 report, London, 69.

COOLEN, J.W., LENGKEEK, W., LEWIS, G., BOS, O.G., VAN WALRAVEN, L. AND VAN DONGEN, U., 2015. First record of Caryophyllia smithii in the central southern North Sea: artificial reefs affect range extensions of sessile benthic species. Marine Biodiversity Records, 8.

FABI, G., SPAGNOLO, A., BELLAN-SANTINI, D., CHARBONNEL, E., ÇIÇEK, B.A., GOUTAYER GARCÍA, J.J., JENSEN, A.C., KALLIANIOTIS, A. AND NEVES DOS SANTOS, M., 2011. Overview on artificial reefs in Europe. Brazilian Journal of Oceanography, 59, pp: 155-166.

FIRTH LB, THOMPSON RC, ABBIAT M, AND AIROLDI, L., 2014. Between a rock and a hard place: environmental and engineering considerations when designing coastal defence structures. Coast Eng 87: 122–35.

FIRTH, L.B., BROWNE, K.A., KNIGHTS, A.M., HAWKINS, S.J., AND NASH, R. 2016. Ecoengineered rock pools: a concrete solution to biodiversity loss and urban sprawl in the marine environment. Environmental Research Letters, Volume 11 (Number 9).

GLAROU, M., ZRUST, M. AND SVENDSEN, J.C., 2020. Using artificial-reef knowledge to enhance the ecological function of offshore wind turbine foundations: Implications for fish abundance and diversity. Journal of Marine Science and Engineering, 8(5), p.332.

HERBERT, R.J., COLLINS, K., MALLINSON, J., HALL, A.E., PEGG, J., ROSS, K., CLARKE, L. AND CLEMENTS, T., 2017. Epibenthic and mobile species colonisation of a geotextile artificial surf reef on the south coast of England. PloS one, 12(9), p.e0184100.

HISCOCK, K., SHARROCK, S., HIGHFIELD, J. AND SNELLING, D., 2010. Colonization of an artificial reef in south-west England—ex-HMS ‘Scylla’. Journal of the Marine Biological Association of the United Kingdom, 90(1), pp.69-94.

JENSEN, A., COLLINS, K. AND SMITH, P., 2000. The Poole Bay artificial reef project. In Artificial Reefs in European Seas (pp. 263-287). Springer, Dordrecht.

JENSEN, A. C. 2002. Artificial reefs of Europe: perspective and future. – ICES Journal of Marine Science, 59: S3–S13

MORRIS, R. L., PORTER, A. G., FIGUEIRA, W. F., COLEMAN, R. A., FOBERT, E. K., AND FERRARI, R. 2018. Fish-smart seawalls: a decision tool for adaptive management of marine infrastructure. Front. Ecol. Environ. 16, 278–287. doi: 10.1002/fee.1809

MOSCHELLA PS, ABBIATI M, ÅBERG P, AND AIROLDI, L., 2005. Low-crested coastal defence structures as artificial habitats for marine life: using ecological criteria in design. Coastal Eng 52: 1053–71.

NAYLOR, L.A., KIPPEN, H, COOMBES, M.A., HORTON, B., MACARTHUR, M. AND JACKSON, N. 2017. Greening the Grey: a framework for integrated green grey infrastructure (IGGI) - Appendix 4: Coastal. Glasgow: University of Glasgow, p56. Available at: http://eprints.gla.ac.uk/150672/

Naylor, L.A., Venn, O., Coombes, M.A., Jackson, J. Thompson, R.C. 2011. Including Ecological Enhancements in the Planning, Design and Construction of Hard Coastal Structures: A process guide. Report to the Environment Agency (PID 110461). University of Exeter, p66.

O’SHAUGHNESSY, K.A., HAWKINS, S.J., EVANS, A.J., HANLEY, M.E., LUNT, P., THOMPSON, R.C., FRANCIS, R.A., HOGGART, S.P., MOORE, P.J., IGLESIAS, G. AND SIMMONDS, D., 2020. Design catalogue for eco-engineering of coastal artificial structures: a multifunctional approach for stakeholders and end-users. Urban Ecosystems, 23(2), pp.431-443.

SAYER, M. D. J., AND WILDING, T. A. 2002. Planning, licensing, and stakeholder consultation in an artificial reef development: the Loch Linnhe reef, a case study. ICES Journal of Marine Science, 59: S178–S185

SHEEHAN, E.V., CARTWRIGHT, A.Y., WITT, M.J., ATTRILL, M.J., VURAL, M. AND HOLMES, L.A., 2020. Development of epibenthic assemblages on artificial habitat associated with marine renewable infrastructure. ICES Journal of Marine Science, 77(3), pp.1178-1189.

TODD, C.D., M.G. BENTLY AND J. KINNEAR. 1992. Torness artificial reef project. In Baine, M.S.P. (ed.) Proceedings of 1st British Conference on Artificial Reefs and Restocking. September 1992, Orkney, ICIT, Stromness,. 15–21. (reference only, cited from Jensen, 2002).

WIGHTLINK 2018. Wightlink's vertipools amaze marine biologists - Wightlink Ferries

WILDING, T. A., AND SAYER, M. D. J. 2002a. Evaluating artificial reef performance: approaches to pre- and post-deployment research. ICES Journal of Marine Science, 59: S222–S230.

WILDING, T. A., AND SAYER, M. D. J. 2002b. The physical and chemical performance of artificial reef blocks made using quarry by-products. ICES Journal of Marine Science, 59: S250–S257.

Appendix 7 Blue mussel: Mytilus edulis

Commercial mussel operations include relaying of juvenile mussels (spat). Translocation is therefore an established technique ad has been used for centuries (Capelle and others, 2014). However, restoring mussel beds in areas where they have been removed can be a challenge as be removal alters hydrodynamics. Restoring populations may therefore require actions to restore habitat suitability. The recovery options discussed in this section include:

Ecoengineering approaches to support recovery

- Artificial reefs (coir rope) (lowered wave and currents, enhanced settlement and reduced predation)
- Predator exclusion (fences to exclude predatory crabs, Shotanus and others 2020)
- Reduction in wave action (Shotanus and others 2020)

Translocation of juvenile mussels (mussel seed, mussel spat). In on-bottom mussel culture, mussel seed dredged from natural seedbeds or collected from spat mussel collectors rope, plastic mesh, etc. attached to a long line and hung in the water column), is positioned at intertidal or sub-tidal lease sites. Commercial beds are laid at high tide by mussel vessels which flush the seed (juvenile mussels) through shafts below water level (seeding). While seeding, the vessel moves in circular patterns. As a result mussels are distributed on multiple plots in concentric patterns. Seeding of high biomasses (up to 150 metric tonnes) is done as fast as possible around slack tide, to prevent mussels flushing out from the lease site by tidal currents. Substantial losses occur in the period after relaying.

Artificial beds

Attachment substrates have been trialled, including coir-nets, and oyster shells in the intertidal.

Coir (coconut) fibre-nets as an attachment substrate for mussels to decrease the chance of getting dislodged were used in a large restoration project of intertidal mussel beds at three lower shore sites (inundation time varied between 85-70%) in the Dutch Wadden Sea (de Paoli and others 2015). The coir net substrates mimic mason worm beds, which often provide a natural substrate for mussel spat to attach to (de Paoli and others, 2015). Subtidal mussels (36,000 kg of adult mussels, size: 5.4 cm ± 6 cm) were obtained by mechanical dredging from a two-year-old natural subtidal mussel bed and manually placed on the artificial beds although the coir net had been rapidly buried underneath 2–3 cm of sediment prior to placement of mussels on the plot, likely due to lugworm activity and sedimentation. All the artificial beds (coir-net and bare sediment controls) disappeared within 200 days. Over all locations mussel loss appeared to result from hydrodynamic forces acting on the bed edges gradually eroded individual mussels from the patches eventually resulting in collapse.

Coir nets were also deployed at an eroding and wave-exposed intertidal mudflat at Viane, in the Oosterschelde estuary (Schotanus and others, 2020). Recruitment to these was compared to a naturally formed oyster (Crassostrea gigas) bed. This species is an invasive non-native species, living oysters were removed or destroyed by breaking their shells 6 weeks before the actual start of the experiment. As a result, the top layer of the oyster beds mostly consisted of oyster shell fragments providing a rough, stable, and complex structure.

Further experiments have been carried out in the Wadden Sea including the Oosterschelde estuary, where mussel bed initiation was tested in a 2 year study by Temmink and others (2022) at an intertidal site. Eight 20x 10m test plots were created, each consisting of 50 modules of artificial establishment structures and coir rope (settlement cue). Establishment structures strongly facilitated reef development on the intertidal flats, while in the bare control sites no reefs developed. Over two years, coverage of the artificial structures declined by approximately 75% through burial and loss.

Multi-factor restoration: Reduction in pressures (predators and wave action and testing of substrate (coir and oysters)

At Viane, in the Oosterschelde estuary (Schotanus and others, 2020) trialled an approach that reduced factors identified as limiting mussel bed assisted recovery (predation and wave action).

Predator exclusion fences

Anti-predator fences to keep out crabs were used to enclose mussel plots of 5 × 5 m with a distance of 1 m from the edge, leaving a buffer zone of 1 m. Fences were made out of 50 cm high plastic mesh with a mesh size of 12 mm, attached between wooden poles 120 cm long that were drilled approximately 80 cm into the sediment. To prevent crabs from climbing over the fence, the top was curved into a U-shape with a diameter of 10 cm. In addition, the fences were dug 10 cm into the sediment to prevent crabs from digging underneath. Before transplantation of the mussels, all plots were searched thoroughly and any crabs present were removed. After six days in the field, mussel coverage in bare sediment plots remained 33% higher when protected by a fence.

Reduction in wave action

Brush wooden dam breakwaters were constructed in front of 12 experimental plots. Breakwaters were 50 cm high, 30 cm wide, and placed 2 m in front of the mussel plots facing southwest. The breakwaters extended 5 m further from the sides of the outermost mussel bed to prevent edge effects.

The complex attachment substrate, (coir-net or oyster shells), substantially increased the retention of transplanted mussel seed substantially and was more effective than changing local hydrodynamic conditions in order to increase survival of transplanted mussels. However mussel losses were still very high with a survival rate of only 29% (Schotanus and others, 2020).

EUNIS Habitats recovery option is applicable to:

- A3.361 *Mytilus edulis* beds on reduced salinity infralittoral rock
- A5.625 *Mytilus edulis* beds on sublittoral sediment

Approaches are not considered feasible to restore A4.241 *Mytilus edulis* beds with hydroids and ascidians on tide-swept exposed to moderately wave-exposed circalittoral rock due to exposure to wave energy that would remove artificial substratum or relaid mussels.

Natural recovery summary for applicable habitats:

The recovery information provided here is taken from the MarLIN sensitivity assessment for A2.72 Littoral mussel beds on sediment (Tillin and Mainwaring, 2015) and that assessment provides further detail.

Recruitment in many *Mytilus* sp. populations is sporadic, with unpredictable pulses of recruitment (Seed and Suchanek, 1992) but larval dispersal is potentially high. Settlement occurs in two phases, an initial attachment using their foot (the pediveliger stage) and then a second attachment by the byssus thread before which they may alter their location to a more favourable one. The final settlement often occurs around or between individual mussels of an established population. In areas of high water flow the mussel bed will rely on recruitment from other populations as larvae will be swept away and therefore recovery will depend on recruitment from elsewhere.

Larval mortality can be as high as 99% due to adverse environmental conditions, especially temperature, inadequate food supply (fluctuations in phytoplankton populations), inhalation by suspension feeding adult mytilids, difficulty in finding suitable substrata and predation (Lutz and Kennish 1992). After settlement the larvae and juveniles are subject to high levels of predation as well as dislodgement from waves and sand abrasion depending on the area of settlement.

In the northern Wadden Sea, strong year classes (resulting from a good recruitment episode) that lead to rejuvenation of blue mussel beds are rare, and usually follow severe winters, even though mussel spawning and settlement are extended and occur throughout the year (Diederich, 2005). In the List tidal basin (northern Wadden Sea) a mass recruitment of mussels occurred in 1996 but had not been repeated by 2003 (the date of the study), i.e. for seven years (Diederich, 2005).

The evidence for recovery rates of M. edulis beds from different levels of impact is very limited and whether these rates are similar, or not, between biotopes is largely unclear. Recovery rates are clearly determined by a range of factors such as degree of impact, season of impact, larval supply and local environmental factors including hydrodynamics.

Overview of approach advantages when compared to natural recovery

In areas where mussel beds have been removed recovery may be inhibited by loss of larval supply, loss of adults to provide suitable substratum for settlement and negative feedbacks around wave exposure that prevent recruitment. Recovery options provide an opportunity to support recovery where this is unlikely to occur naturally.

Overview of approach disadvantages compared to natural recovery

Assisted recovery may have high failure rates. Mussel bed restoration in the Wadden Sea (Paoli 2015) study results revealed a near disappearance of all experimental beds in just over 7 months .These findings highlight that restoration of beds in dynamic areas cannot be implemented by mussel transplantation alone when other processes are influencing the loss of mussel beds. However, the study by Schotanus and others (2020) also experienced high losses despite mitigating for limiting factors.

Blue mussel assisted recovery: economic costs

The costs to restore intertidal mussel beds was calculated based on artificial structures and rope to range from USD $106,00 to $318,000 (£81,297-£243,893/ha based on current exchange rate) depending on plot density (Temmink and others 2021). These approaches are relevant only to intertidal plots and a further breakdown of these is not provided.

Although a cost was not presented, Schotanus and others (2020) stated that while the fences used in their project were very effective in lowering establishment thresholds, but they were also labour intensive and costly and therefore inadvisable for large-scale restorations.

No costs could be identified for dredging and relaying mussel spat. This was considered to be a relatively low cost option although, outside of established aquaculture areas, projects should allocate costs for licensing and feasibility assessments, given the high failure rates . It is not clear how much predator management is typically carried out (for example the use of starfish mops ‘’) and this is probably site specific.

Blue mussel assisted recovery: Feasibility

Artificial structures were successful at initiating *M. edulis* recruitment but artificial structures suffered high losses (Temmink and others, 2022). Generally, wild seed collection only occurs in areas where the accumulation of mussel seed would not persist naturally; so called ‘ephemeral’ accumulations (SEAFISH). Even in favourable conditions losses and predation rates may be high (Capelle and others, 2014). In sites outside of very sheltered conditions, establishment of beds may be constrained by negative feedbacks. Aquaculture areas in sheltered locations are characterised by sediments, the applicability of this approach to mussel beds on rock is therefore unclear. The intertidal approaches are discussed as they may be applicable to the sublittoral fringe but are unlikely to be used to support subtidal recovery.

Blue mussel assisted recovery: risks, challenges and uncertainties

Risks. High failure rates: even within sheltered sites selected for aquaculture, high losses are typical, with up to 75% loss in the first month (Capelle and others, 2014). Mussel beds thrive in high energy environments where water movement provides food. However, these conditions can also inhibit recruitment. On wave-exposed mudflats, transplanted mussels may not be able to form byssus attachments and large groups before they are washed away or eaten. The experiments by Temmink et al (2022) indicate high losses of structures through burial and loss. Experimental beds on the lower shore nearly all disappeared beds in just over 7 months restoration of mussel beds in dynamic areas cannot be achieved by mussel transplantation alone (de Paoli and others 2015).

Blue mussel assisted recovery: references

CAPELLE, J.J., WIJSMAN, J.W., SCHELLEKENS, T., VAN STRALEN, M.R., HERMAN, P.M. AND SMAAL, A.C. 2014. Spatial organisation and biomass development after relaying of mussel seed. Journal of Sea Research, 85, pp.395-403.

CALDERWOOD, J., O'CONNOR, N.E. AND ROBERTS, D. 2016. Efficiency of starfish mopping in reducing predation on cultivated benthic mussels (Mytilus edulis Linnaeus). Aquaculture, 452, pp.88-96.

DE PAOLI, H., VAN DE KOPPEL, J., VAN DER ZEE, E., KANGERI, A., VAN BELZEN, J., HOLTHUIJSEN, S., VAN DEN BERG, A., HERMAN, P., OLFF, H. AND VAN DER HEIDE, T. 2015. Processes limiting mussel bed restoration in the Wadden-Sea. Journal of Sea Research, 103, pp.42-49.

DIEDERICH, S. 2005. Differential recruitment of introduced Pacific oysters and native mussels at the North Sea coast: coexistence possible? Journal of Sea Research, 53 (4), 269-281.

LUTZ, R.A. AND KENNISH, M.J., 1992. Ecology and morphology of larval and early postlarval mussels. The mussel Mytilus: ecology, physiology, genetics and culture. Elsevier, Amsterdam, pp.53-85.

SEED, R. AND SUCHANEK, T.H., 1992. Population and community ecology of Mytilus. In The mussel Mytilus: ecology, physiology, genetics and culture, (ed. E.M. Gosling), pp. 87-169. Amsterdam: Elsevier Science Publ. [Developments in Aquaculture and Fisheries Science, no. 25.]

SCHOTANUS, J., CAPELLE, J.J., PAREE, E., FIVASH, G.S., VAN DE KOPPEL, J. AND BOUMA, T.J. 2020. Restoring mussel beds in highly dynamic environments by lowering environmental stressors. Restoration Ecology, 28(5), pp.1124-1134.

TEMMINK, R.J., ANGELINI, C., FIVASH, G.S., SWART, L., NOUTA, R., TEUNIS, M., LENGKEEK, W., DIDDEREN, K., LAMERS, L.P., BOUMA, T.J. AND VAN DER HEIDE, T., 2021. Life cycle informed restoration: Engineering settlement substrate material characteristics and structural complexity for reef formation. Journal of Applied Ecology, 58(10), 2158-2170.

TEMMINK, R.J., FIVASH, G.S., GOVERS, L.L., NAUTA, J., MARIN-DIAZ, B., CRUIJSEN, P.M., DIDDEREN, K., PENNING, E., OLFF, H., HEUSINKVELD, J.H. AND LAMERS, L.P. 2022. Initiating and upscaling mussel reef establishment with life cycle informed restoration: Successes and future challenges. Ecological Engineering, 175, p.106496.

TEMMINK, R.J., ANGELINI, C., FIVASH, G.S., SWART, L., NOUTA, R., TEUNIS, M., LENGKEEK, W., DIDDEREN, K., LAMERS, L.P., BOUMA, T.J. AND VAN DER HEIDE, T., 2021. Life cycle informed restoration: Engineering settlement substrate material characteristics and structural complexity for reef formation. Journal of Applied Ecology, 58(10), pp.2158-2170.

TILLIN, H.M. AND MAINWARING, K., 2015. Littoral mussel beds on sediment. In Tyler-Walters H. Marine Life Information Network: Biology and Sensitivity Key Information Reviews, [on-line]. Plymouth: Marine Biological Association of the United Kingdom. [cited 13-03-2022]. Available from: https://www.marlin.ac.uk/habitat/detail/299

Appendix 8 Native oyster

Techniques for the assisted restoration of native oyster reef/bed focus on either supporting larval supply through restocking or substratum restoration to support settlement. Depending on site-specific factors limiting natural recovery, one or both approaches may be required.

Native oyster restocking through reseeding or relaying individuals or spat on shell.

Restocking to enhance brood stock increases the population of native oyster for the purpose of reproduction and larval supply. Juvenile oysters (spat) may be sourced from hatcheries or collected from the wild. Helmer and Hancock (2020) discuss the benefits and limitations of each of these approaches. An oyster hatchery is a facility where adults are conditioned to reproduce and spawn, and larvae are reared until metamorphosis (with or without settlement). Spatting ponds may be used. These are large pits that are filled with seawater. Mature oysters are placed in the ponds during the spawning season and as soon as settled spat are observed settlement material (shell) is added to the ponds. Traditionally bivalve shells, in particular mussel, scallop and oyster shells are used, with the opinion amongst producers that mussel shell obtains the greatest settlement (Helmer and Hancock, 2020). However, for restoration, scallop and oyster shells may provide a more suitable solution as they are heavier and therefore less likely to be removed by tides and currents when placed onto the seabed or prepared reefs.

A range of spat collectors have been trialled including shell cultch, bundles of twigs, limed tiles and commercially made collectors. If available, hydrodynamic models within the local system can provide information regarding possible recruitment hotspots and inform the placement of collectors. Any removal of oysters from the seabed should take place in accordance with biosecurity measures, with regulations and bylaws put in place by the local fisheries authority, (the Inshore Fishery and Conservation Authority (IFCA) in England.

Single juvenile oysters, not attached to a substratum (cultchless spat), can be produced by removal from initial settlement surfaces. Removing the settlement substratum removes associated mass and predator protection, making the oysters prone to unintended translocation in tides and currents and cultch-less spat are likely to be subject to higher predation rates.

Deploying mature adult oysters is particularly key in recruitment limited environments as they can begin to provide an immediate supply of larval output to address this issue. It may be possible to purchase mature 'brood stock' oysters (> 50mm), where there is a sustainable oyster fishery, to be translocated into an area that is protected from fishing pressures and make biosecure in large quantities providing a ready-grown source of potential larvae. Mature brood stock oysters offer the potential of spawning and larval output during the first year of deployment, have a good survival rate, and may provide chemical cues for the settlement of 'wild' spat in the system.

Oyster deployment: The simplest way to reseed single oysters, or spat-on-shell, is from a vessel with appropriate lifting gear. With an experienced skipper a vessel can run several transects up and down the selected relaying area at a set speed. The number of oysters to be deployed from the vessel for each transect or 'dump' can then be calculated. For example, if the boat was travelling at 1 m per second, one oyster would be dropped every second along the transect. This would achieve reef density of 1 oyster per m^2.

Native oyster substratum enhancement

Although existing adults are the best substratum for settlement (see recovery below), dead shell is also a suitable substratum for settlement. Although *O. edulis* has been shown to settle on a variety of substrata, including hard silt, muddy gravel with shells, sand and rocks, larvae favour other oyster shells. Therefore, brood stock oysters are used to attract larvae in restoration projects. For areas of poor quality subtidal mixed sediment the addition of large quantities of shell or alternative material are required to stabilise sediments, provide reef foundations, allow for settlement of larvae and prevent dispersal of oysters deployed in the area. The shell available in an area can be determined when conducting initial benthic surveys. As reefs establish, there will be both gain and loss of shell material through natural processes, but ultimately the gain needs to exceed the rate of loss.

Deployment of stone aggregates or various shell types (cultch) offers settlement material in a substratum limited environment, where larval supply is not necessarily a limiting factor. Cheaper materials such as stone and gravel aggregates can raise the height from the seabed before higher cost shells and oysters are deposited on top (Helmer and others, 2020). Individual requirements for gravel or shell type will differ with local regulations. Raising oysters off the seabed reduces the effects of sediment smothering and associated mortality, as well as improving their physiological performance.

In the Blackwater Estuary, the oyster grounds are improved by harrowing (a process of dredging sites in summer to disturb and redistribute bottom sediments and *Crepidula fornicata* colonies, and re-expose shell cultch (Bromley and others, 2016), and relaying shell cultch in large quantities (cited from Allison and others 2019).

Small-scale on and off bottom techniques.

Small-scale methods to introduce oysters can be used as pilot studies, to engage communities and to complement larger scale reef restoration by providing additional larval supply as well as some form of protection. 'On-bottom' describes techniques that involve depositing cage or concrete structures directly onto the seabed. 'Off-bottom methods' describe techniques that involve suspending oysters above the seabed in cages, floating oyster systems or long-lines.

EUNIS Habitats recovery option is applicable for:

EUNIS A5.435 *Ostrea edulis* beds

Native oyster - natural recovery summary:

The native oyster settles in groups, preferring to settle on an adult of the same species, resulting in layers of oysters. Adults are cemented to the substratum. Adult immigration is not possible and recovery is dependent on the larval phase. This species can be highly fecund, producing an average of between 91,000 to up to 2 million eggs. A number that increases with age and size. However, good fertilization efficiency requires a minimum population size, so that in small populations not all the eggs may be fertilized (Spärck, 1951). The size of the sexually mature population and the production of larvae are not accurate ways of predicting the success of spatfall (Gravestock and others, 2014). The larvae are pelagic for 11-30 days, providing potentially high levels of dispersal, depending on the local hydrographic regime. In areas of strong currents, larvae may be swept away from the adult populations to other oyster beds. Oyster beds on open coasts may be dependent on recruitment from other areas, while oyster beds in enclosed embayments may be self-recruiting. Due to the high numbers of larvae produced, a single good recruitment event could potentially significantly increase the population. However, recruitment in *O. edulis* is sporadic and dependent on local environmental conditions, including the average summer seawater temperature, predation intensity and the hydrographic regime.

The main determinants of larval settlement are substratum availability, adult abundance, and local environmental conditions and hydrographic regime (Preston and others, 2020). Oyster settlement is known to be highly sporadic, and spat can suffer mortality of up to 90% (Cole, 1951). This mortality is due to factors including, but not restricted to; temperature, food availability, suitable settlement areas, and the presence of predators (Cole, 1951; Spärck, 1951; Kennedy and Roberts, 1999; Lancaster, 2014). Populations undergo natural phases of expansion and contraction. Successful recruitment appears to vary between one to three years (Loch Ryan, Scotland), or even every 6-8 years (Lough Foyle) (MMO, 2019).

Larvae respond to environmental cues that guide them to settling within the most suitable locations (Walne, 1974; Woolmer and others, 2011). Bayne (1969) stated that *O. edulis* larvae are highly gregarious and will preferably settle where larvae have previously settled. A number of other studies have also found that larvae preferentially select well-stocked beds to degraded beds or barren sediment (Cole and Knight-Jones, 1939, 1949; Walne, 1964; Jackson and Wilding 2009; cited in Gravestock, 2014). In addition to live settled oysters, spat will also settle selectively on recently dead oysters Woolmer and others, (2011) and oyster cultch (shell) (Kennedy and Roberts, 1999). Other bivalve cultch can also encourage settlement of oyster spat, although which species of shell is most beneficial to this is debated (Gravestock and others, 2014).

Connectivity with naturally occurring brood stock is desirable as a potential source of larvae. Connectivity may also contribute to increased resilience of both the restored and existing habitats. Hydrodynamic studies are important for understanding patterns of water flow and hence larval connectivity.

Regime shifts: Oyster beds may be replaced by deposit feeding polychaetes that may influence the recovery of suspension feeding species. Following the reduction in oyster populations, re-establishment can be restricted by invasive non-native species. The slipper limpet, *Crepidula fornicata*, can become dominant in oyster habitat and restrict recovery through changes to the environment and competition (Blanchard, 1997; Hawkins and others, 2005; Laing and others, 2006 cited in Gravestock and others, 2014).

Predation by invasive species: Newly settled spat and juveniles are subject to intense mortality due to predation, especially by the oyster drills (*Urosalpinx cinerea* an invasive non-native species, and *Ocenebra erinacea*) and starfish. For example, in the Oosterschelde, Korringa (1952) reported 90% mortality in oyster spat by their first winter, with up to 75% being taken by *U. cinerea*, while Hancock (1955) noted that 73% of spat settling in summer 1953 died by December, 55 -58% being taken by *U. cinerea*.

Competition for space (substratum for settlement) from other species that settle at the same time of year for example, barnacles and ascidians, results in high levels of larval and juvenile mortality.

Overview of approach advantages when compared to natural recovery

Native oyster reefs are now among the most threatened marine habitats in Europe. In the UK and Ireland populations have declined by 95%, with remnant populations found in the south east of England, west coast of Scotland and the south coast of Ireland. Features that may limit recoverability in *O. edulis* include 1) brood stock size, 2) decrease in preferred settlement surfaces (*O. edulis* beds and shell), and 3) adult mortality (disease and harvesting). The significant decline of native oyster populations across Europe has highlighted that active intervention is required for the recovery of this species (Preston and others, 2020).

Overview of approach disadvantages compared to natural recovery

Potential disadvantages that should be considered include impacts on donor population and biosecurity risks from invasive non-native species. Oyster supply is a key limiting step in oyster restoration projects. Sourcing oysters from outside the local area can present significant biosecurity risks, which are time consuming and costly to address. Potential impacts on the donor site must be considered when considering using wild stocks,.

Native oyster assisted recovery: economic costs

Options for oyster reef material and placement costs are provided in Table 17 along with a range of estimates for seed costs, with costs from Hynes and others (2022). Costs were adjusted from that paper based on an exchange rate of 1 GBP = 1.18874 EUR, no adjustment was made for the 2018 costs as reported in Hynes and others (2022).

Capital costs

It is generally not possible to guarantee the origin of shell accessed for restoration. Therefore, all shell is considered a potential biosecurity risk from a separate water body. As a result, all shell needs to be cleaned and cured to remove all biological material and any potential pathogens.

Shell recycling schemes utilise the ‘biological waste material’ produced by shellfish consumption in the restaurant industry and are conducted on an enormous scale by numerous restoration projects in the United States. They can provide an opportunity for community engagement and reduce landfill. Shell will still need to be assessed and processed according to biosecurity measures.

Seed costs are based on estimated from Laing and others (2006). In particular the cost of *O. edulis* adult seed was based on those reported for the Essex reef restoration project (Essex Native Oyster Initiative). All figures were adjusted from original years to €2018 values using the relevant industrial input price index (or agricultural input price in the case of seed costs), exchange rate and purchasing power parity by Hynes and others (2022). These costs are shown in Table 17 and converted to GBP.

Operational costs

For oysters suspended in cages, regular cleaning ensures that species that compete for settlement area are kept at manageable levels. Biosecurity risks can be costly and time consuming to deal with

Hynes and others (2022) calculated the costs associated with putting in place an oyster reef bar consist of purchasing reef material, placing the material and the costs of seeds. The dimensions of the reef bar are assumed to be 1070 m by 6 m by 1 m. It was assumed a pacific oyster shell substratum is used, supported by mesh. A seeding rate of 100,000 half-grown native oysters per hectare was also assumed. Cost estimates of reef material (substratum) purchase and placement were based on a number of international oyster reef restoration initiatives and their associated construction/restoration costs.

Maintenance costs

It was further assumed that monitoring would be carried out by two scientific staff (assuming one senior and one post-doctoral level scientist) five days per year based on Laing and others (2006). Maintenance of the ground (cleaning the ground of potential predators, removal of litter, etc.) is necessary and it is assumed it would be done during the summer (when spawning occurs) using relatively cheap methods such as mops or lines over a period of 10 working days. This amounts to annual monitoring and maintenance labour costs of €12,640. An additional €2,500 is assumed for monitoring and maintenance equipment costs per annum from year 1 onwards.

Table 17. Oyster reef assisted recovery costs from Hynes and others 2022.

Project	Description	Costs	Reference
Renville Bay	Maintenance cost: Monitoring and maintenance (2 scientific staff per annum- 1 senior, 1 post-doc 5 days/year	0.64 ha: Staff €12,649, equipment, €2,500 (*Total: £12,748, 1ha= £19,919)	Hynes and others, 2022
Renville Bay (total cost)	Cost of reef material (substrate) purchase and placement with mesh	0.64 ha: €246,000 (*£206,977) 1 ha: €384,375 (*£323,432)	Hynes and others, 2022
-	Seed from ponds/hatcheries at 20-40mm	€5,850/ha (£*4,920)	Laing and others, 2006; cited in Hynes and others, 2022
-	Bonamia free area, half grown native oyster	€1280-1800/ha (£*1,077-	
Essex Native Oyster Initiative	Reef material (shell and cockle), purchase and placement	€183,492/ha (*£154,387/ha)	Hynes and others 2022

Native oyster assisted recovery: benefits

Ecosystem services associated with recovered habitats: Oysters are ecosystem engineers. Restoring oysters increases biodiversity, through the existence of oysters themselves, and the biodiverse reef system they create. Healthy shellfish habitat can also positively benefit associated habitats such as seagrasses (Sharma and others, 2016).

Coastal protection: As a natural coastal defence alternative oyster reefs can function as natural breakwaters as they interact with tidal and wave energy to reduce shoreline erosion. A number of studies have demonstrated that oyster reef restoration can provide significant shoreline protection (see Hynes and others 2022). Oyster reefs also have the added advantage that they automatically adjust to sea level rise as they can grow vertically faster than sea levels are expected to rise.

Nursery habitats: Oyster reefs can increase fish production by providing a protective nursery ground for juveniles that acts as a refuge from predation and provides a source of food through increasing the abundance of prey.

Carbon sequestration: Through biodeposition and passive sedimentation carbon may be stabilised, and along with shell assimilation, integrated into the oyster reef as it grows over time. Further to these real-time factors a number of long-term processes must also be considered such as erosion, microbial activity and bioturbation (Preston and others 2020). The Dorncoh Firth DEEP Project is studying carbon sequestration by the restored oyster reef.

Waste remediation/ filtration: A single oyster can filter up to 200 litres of seawater per day, which can significantly improve water quality and clarity. Oysters can also assimilate excess nutrients and promote microbial activity in the underlying sediments to denitrify nitrates and nitrites, thus removing them from the water body. Oyster restoration in the Dornoch Firth was funded to support organic waste remediation.

Economic benefits: Fisheries: Restored beds could support fisheries if sustainable managed. Oyster densities in areas new to fishery exploitation appear to have been high. In the 1780s, the 20 mile long reef in the Firth of Forth, Scotland, was estimated to produce as many as 30 million oysters per year, employing up to 60 boats each manned by five workers (Preston and others, 2020). Protected restoration areas can provide spill-over of larvae that may seed and support sustainable fisheries.

Economic benefits: job creation Restoration activities including production or procurement of oysters, the placement of substratum for oyster settlement, project management and monitoring create jobs and generate employment which benefits the local economy. Selecting a location that could benefit significantly from such investment may be a consideration in restoration projects with a strong socioeconomic focus.

Native oyster assisted recovery: feasibility

Native oyster stock restoration is feasible, especially in disease free areas. A number of groups have been established and technical guidelines produced to support Native oyster restoration. Those relevant to the UK include:

1) The European Native Oyster Restoration Alliance (NORA) was established during an international workshop on native oyster restoration hosted by the German Federal Agency for Nature Conservation (BfN) and the Alfred Wegener institute (AWI) in Berlin in November

2017. During this workshop, key issues for successful Europe wide restoration were identified and summarised in the “Berlin Oyster Recommendation”. Since this inaugural workshop, NORA has hosted conferences and set up working groups to address key topics in restoration practice and bottlenecks to scaling such as: site selection, biosecurity, monitoring and oyster production.
2) The Native Oyster Network is a community of academics, conservationists’ oyster-persons and NGOs who are working together to restore self-sustaining populations of native oysters established in 2017 by the Zoological Society of London and University of Portsmouth. Website: https://nativeoysternetwork.org/
3) The Environment Agency has developed a GIS Native Oyster Bed Potential Area layer that provides a national ‘high level’ indication of where native oyster reefs could potentially be restored as an initial aid to identifying sites.

Examples of UK Projects

1) In Essex the gravels used to elevate the oysters were required to be of a type naturally occurring in the estuary, which could come from a local land-based gravel pit.
2) Heavy scallop shells were used for experiments in the Dornoch Firth to stabilise the substratum and increase oyster retention in order to establish if shell reefs could be recreated.
3) An industry led restoration project was undertaken in Stanswood Bay, Solent (UK) where cultch and brood-stock oysters were deposited in order to increase the larval supply to the surrounding areas. This attempt was affected by a series of factors including disease and habitat change (Woolmer and others, 2011). Similar restocking schemes have also been undertaken in Chichester Harbour, Falmouth, Carlingford Lough, Spain and the Limfjord in Denmark (Dolmer and Hoffmann 2004 in Woolmer and others, 2011).

Selecting a site that not only supports settlement, but also survival, growth and reproduction is fundamental in the long-term success of any restoration project (see site selection guidelines by Hughes and Zu Ermgassen, 2021). Sites should have the following characteristics:

1) Absence of threats that threaten oyster populations
2) Suitable environmental conditions: low sedimentation rate, low pollution levels, minimal sewage outflow within close proximity, absence of high-impact invasive non-native species; absence of pathogens.
3) Site accessibility: in order to establish and maintain a native oyster habitat restoration site, the area selected must be accessible at important times in the oyster’s life cycle and at regular intervals throughout the year to conduct maintenance and monitoring activities. Travel time, staff safety, water depth, wave height and storm frequency may all be factors which affect site accessibility both inshore and offshore, as well as the associated costs of access and monitoring.
4) Spatial scale: a common aim of native oyster restoration projects is to establish a self-sustaining population. Given the interannual variability in recruitment, the influence of tides and currents, and the sessile nature of adults, oyster population sustainability is likely to be achieved at larger rather than smaller scales. Restoration sites should, therefore, consider whether there is sufficient suitable habitat surrounding the restoration site to accommodate population expansion and build up a resilient native oyster population.

Spat collectors or small-scale pilots can be deployed to obtain a measure of settlement rates or alternatively plankton surveys may be undertaken. If larvae are available in sufficient numbers, they need suitable substrate on which to settle.

Timing of seabed deployment of oysters or substrate is a critical factor. Placing large amounts of juvenile oysters on the seabed at times of the year when predators, such as crabs (for example, *Carcinus maenas*), are in high abundance in coastal waters will result in unnecessary mortality. When deploying substratum, doing so too early can result in algal turf and other organisms settling before the oysters begin to search for suitable substrate and metamorphose. Alternatively, deploying too late in the season won't allow for sufficient biofilm formation and will mean the larvae are likely to either disperse and settle elsewhere, or not settle at all due to a lack of suitable substratum. Numerous factors, including temperature, lunar cycle and food availability influence the timing, health and quantity of larvae released by female oysters, but an indication of the peak in activity can be observed from previous documentation and comparing that with current observations. This is likely to vary across the biogeographic range of the native oyster, as well as locally with changes in climatic conditions. It is recommended that larval abundance surveys be conducted in the intended area to be restored, at least for the season prior to deployment of larger scale aspects of the project.

Native oyster assisted recovery: risks, challenges and uncertainties

Challenges: Oyster supply is a key limiting step in oyster restoration projects. Sourcing oysters from outside the local area can present significant biosecurity risks, which are time consuming and costly to address and impossible to eliminate completely. When considering using wild stocks, the impact on the donor site must be considered first. The use of wild stocks to supply the demand from restoration has the potential to further damage the remaining populations. Projects must ensure that the stock selection process is conducted responsibly and in accordance with legislation and biosecurity protocols (Helmer and Hancock, 2020). When sourcing stock from a hatchery, establish whether the seed supplied has been in open contact with the surrounding water body before being shipped. If so, biosecurity protocols equivalent to being moved from the open water body must be applied. Alternatively, projects can consider buying biosecure hatchery stock and growing them out locally for 18 months to two years (Helmer and Hancock, 2020).

Risks: Biosecurity. Restoration projects should perform appropriate risk assessments of their activities with biosecurity in mind, and that protocols are developed to minimise risks where they are identified. The most impactful pathogens affecting oysters are those identified by the World Organization for Animal Health (OIE) and/or the European Commission (EC). These diseases are *Bonamia ostreae*, *B. exitiosa*, *Marteilia refringens*, *Mikrocytos mackini*, and *Herpes* virus OsHV-1μVar. All pose serious biosecurity threats. Disease status has far reaching consequences for biosecurity protocols and the potential survival of naive oysters relayed to the area (Preston and others, 2020 and zu Ermgassen and others, 2020).

Risks: using wild caught oysters. Movement of such oysters should ideally take place within the same body of water, and if this is not possible, appropriate biosecurity risk assessment and practice should be planned into the project timeline and budget. Projects should consider that moving large numbers of oysters may be infinitively costly, time consuming and risky (Helmer and Hancock, 2020).

Risks: biofouling. A high abundance of biofouling organisms can result in native oysters being outcompeted for space on suitable substratum. At locations with a high abundance of biofouling species, careful timing of cultch placement can to some extent mitigate the impact of this competition in the near term, and features of the surrounding environment, such as high kelp cover, can help to shade and suppress species that compete for settlement area (Shelamoff and others, 2019).

Risks: invasive non-native species: The American slipper limpet (*C. fornicata*), carpet sea squirt (*Didemnum vexillum*), and American oyster drill (*U. cinerea*) are particularly damaging to native oyster populations and/or the surrounding benthic community. The presence of high impact INNS at a site does not preclude its restoration. However, consideration should be given as to how the presence of high impact INNS influences biosecurity measures taken by the project, as well as the potential impact on project progress (Zu Ermgassen and others, 2020). Pacific oyster may be beneficial in enhancing recruitment of the native oyster, by providing shell material for settlement and improving structural stability of the reef (Christianen and others, 2018). Biosecurity is a consideration with the laying of cultch, shells weathering on land for 12 months minimises risks posed by potential pathogens and invasive non-native species. Stone used to enhance substrata for settlement requires no weathering if from a land based source

Challenges: maintaining genetic diversity. Translocations of native oyster stock can also have implications for the genetic diversity within the species. Historically there have been many translocations of oysters across the UK and Ireland and a degree of genetic homogeneity already exists. However, studies also demonstrate there is relatively high diversity and geographical differentiation in the genetic population structure across the native oyster's biogeographic range. Genetic differentiation has been linked to both adaptations and disease resilience at local scales. For this reason, it is important that restoration practices, at a minimum, maintain local or regional genetic diversity and adaptations (Preston and others, 2020. In addition, restoration projects should seek to utilise breeding techniques that maximise the genetic diversity in the offspring to enable resilience to future change (Helmer and Hancock, 2020).

Challenges: unregulated harvesting or poaching. At Strangford Lough a decline in *O. edulis* numbers was reported, which was attributed to unregulated fishing activity.

Challenges: historical baselines. The historical decline of the native oyster predates rigorous monitoring and survey. Records of what a 'pristine' native oyster biogenic habitat looked like, how densely oysters were clustered together and the species they supported, are extremely rare for native oysters. Beds would have been subjected to some form of physical alteration decades or even centuries before scientific descriptions took place. The descriptions of density that do exist vary widely and almost certainly reflect impoverished populations (Gamble and others 2020). An average of 1 live oyster per m^2 was recorded in the Fal oyster fishery in 1924, while just 0.001 live oysters per m^2 were recorded in a relic oyster population in northern Strangford Lough, Northern Ireland.

While the historical presence of oysters can be confirmed through fisheries records or shell deposits in many locations, this is not always possible, in particular offshore. The historical range of the native oyster is, however, better understood and can serve as a guide as to whether restoration or reintroduction (as opposed to an introduction) of native oysters is appropriate at a given site. Incorporation of such knowledge may improve the likelihood that a site is suitable for present day restoration efforts. Though physical evidence is desirable, it is not present at a large number of current restoration sites, in particular in offshore waters. In such cases, knowledge from surrounding areas may be useful.

Uncertainties: deployment densities: The end goal of restoration is often a sustainable population, and it is not yet known how this relates to density or area of oyster reef habitats. The initial target density should be informed where possible by historical records, ecological data, and stakeholder input. The latter is likely to be important in particular where restoration efforts are co-located with fisheries, given the potential for oyster density to interact with disease prevalence (Helmer and Hancock, 2020). The density achieved immediately after deploying oysters, especially for spat-on-shell or juveniles, will need to be substantially greater than the intended established density. Surveys in the Solent showed that as few as 5% may be retained after one year when relaying juvenile oysters (25-30 mm in size) directly onto the seabed. Similarly, in the Dornoch Firth, densities of 10-15 g oysters reduced by >50% in three months due to tidal redistribution in a 2 knot tide. It is likely that the use of shell or stone material, to create stable reef structures, can increase the rugosity of the seabed and therefore retention on the target area. Retention of oysters will be different at each project site. Therefore, practitioners are encouraged to run small-scale pilot studies in order to understand the hydrodynamics, rate of retention, predation or mortality for the site and accommodate for losses associated with these issues, feeding the results into potential retention calculations (Helmer and Hancock, 2020).

Uncertainties: bivalve shell selection for spat on shell deployment. The shell material used will depend on availability, site dynamics (wave action, currents etc) and annual settlement of larvae. Trials should be conducted prior to large-scale deployments.

Native oyster assisted recovery: references

ALLISON, S., HARDY, M., HAYWARD, K., CAMERON, T.C. AND UNDERWOOD, G.J. 2020. Strongholds of Ostrea edulis populations in estuaries in Essex, SE England and their association with traditional oyster aquaculture: Evidence to support a MPA designation. Journal of the Marine Biological Association of the United Kingdom, 100(1), pp.27-36.

BLANCHARD, M. 1997. Spread of the slipper limpet Crepidula fornicata (L.1758) in Europe. Current state and consequences. Scientia Marina, 61, Supplement 9, 109-118.Christianen et al 2018

BROMLEY, C, MCGONIGLE, C, ASHTON, EC AND ROBERTS, D. 2016. Restoring degraded European native oyster, *Ostrea edulis*, habitat: is there a case for harrowing? Hydrobiologia 768, 151–165.

CHRISTIANEN, M.J.A., LENGKEEK, W., BERGSMA, J.H., COOLEN, J.W.P., DIDDEREN, K., DORENBOSCH, M., DRIESSEN, F.M.F., KAMERMANS, P., REUCHLIN-HUGENHOLTZ, E., SAS, H. AND SMAAL, A., 2018. Return of the native facilitated by the invasive? Population composition, substrate preferences and epibenthic species richness of a recently discovered shellfish reef with native European flat oysters (Ostrea edulis) in the North Sea. Marine Biology Research, 14(6), pp.590-597.

COLE, H. AND KNIGHT-JONES, E.W. 1939. Some observations and experiments on the setting behaviour of larvae of Ostrea edulis. Journal du Conseil Permanent International pour L'Exploration de la Mer, 14, 86–105.

COLE, H. AND KNIGHT-JONES, E.W. 1949. The setting behaviour of larvae of the European flat oyster, O. edulis L. and its influence on methods of cultivation and spat collection. Ministry of Agriculture, Fisheries and Food, Fisheries Investigations Series II, 17, 1–39.

COLE, H.A. 1951. The British oyster industry and its problems. Rapports and Proces-Verbaux des Reunions. Conseil Permanent International pour l'Exploration de la Mer, 128, 7-17.

GAMBLE, C., ASHTON, E.C., BROMLEY, C., FARIÑAS-FRANCO, J.M., HANCOCK, B., HAYDEN-HUGHES, M., HELMER, L., LEE, H.Z., PRESTON, J., SANDERSON, W.G. AND THURSTAN, R., 2020. European Native Oyster Restoration Handbook: An introduction. In

European Native Oyster Habitat Restoration Handbook: UK & Ireland (pp. 2-11). The Zoological Society of London.

GRAVESTOCK, V., JAMES, F. AND GOULDEN, M. 2014. Solent native oyster (Ostrea edulis) restoration - Literature Review & Feasibility Study. MacAlister Elliot & Partners, On behalf of the Blue Marine Foundation, Report, no. 2897.

HANCOCK, D.A., 1955. The feeding behaviour of starfish on Essex oyster beds. Journal of the Marine Biological Association of the United Kingdom, 34(2), pp.313-331.

HELMER, L., HANCOCK, B., BONACIC, K., BROMLEY, C., FABRA, M., FRANKIĆ, A., HAYDEN-HUGHES, M., HOLBROOK, Z., KEAN-HAMMERSON, J., MCAFEE, D., MOUNTAIN, C., NEDOSYKO, A., PRESTON, J., RODRIGUEZ-PEREZ, A., SANDERSON, W., SMYTH, D., UTTLEY, M., AND ZU ERMGASSEN, P. 2020. Chapter 3 Native oyster restoration in practice In: European Native Oyster Habitat Restoration Handbook (eds. J. Preston, C. Gamble, A. Debney, L. Helmer, B. Hancock and P.S.E. zu Ermgassen), pp.2-11. The Zoological Society of London, London, UK.

HUGHES, A. AND ZU ERMGASSEN, P.S.E. 2021. European Native Oyster Habitat Restoration Site Selection Checklist, Native Oyster Restoration Alliance, Berlin, Germany.

HYNES, S., BURGER, R., TUDELLA, J., NORTON, D. AND CHEN, W. 2022. Estimating the costs and benefits of protecting a coastal amenity from climate change-related hazards: Nature based solutions via oyster reef restoration versus grey infrastructure. Ecological Economics, 194, p.107349.

JACKSON E. AND WILDING, C. 2009. Ostrea edulis. Native oyster. Marine Life Information Network: Biology and Sensitivity Key Information Sub-programme [On line]. Plymouth: Marine Biological Association of the United Kingdom. [cites 03.02.16]. Available from: http://www.marlin.ac.uk/speciesfullreview.php?speciesID=3997

KENNEDY, R.J. AND ROBERTS, D. 1999. A survey of the current status of the flat oyster Ostrea edulis in Strangford Lough, Northern Ireland, with a view to the restoration of its Oyster Beds. Biology and Environment: Proceedings of The Royal Irish Academy, 99B, 79–88.

KENNEDY, R.J. AND ROBERTS, D. 2006. Commercial oyster stocks as a potential source of larvae in the regeneration of Ostrea edulis in Strangford Lough, Northern Ireland. Journal of the Marine Biological Association of the United Kingdom, 86(1), pp.153-159.

KORRINGA, P. 1952. Recent advances in oyster biology. Quarterly Review of Biology, 27, 266-308 & 339-365.

LAING, I., WALKER, P. AND AREAL, F.2006. Aquatic Living Resources Return of the native – is European oyster (Ostrea edulis) stock restoration in the UK feasible?', Aquat. Living Resour. EDP Sciences, 19, pp. 283–287. doi: 10.1051/alr:2006029.

LANCASTER, J. (ED), MCCALLUM, S., A.C., L., TAYLOR, E., A., C. AND POMFRET, J., 2014. Development of Detailed Ecological Guidance to Support the Application of the Scottish MPA Selection Guidelines in Scotland's seas. Scottish Natural Heritage Commissioned Report No.491 (29245), Scottish Natural Heritage, Inverness, 40 pp.

MMO 2019. Identifying sites suitable for marine habitat restoration or creation. A report produced for the Marine Management Organisation by ABPmer and AER, MMO Project No: 1135, February 2019, 93pp

PRESTON J., GAMBLE, C., DEBNEY, A., HELMER, L., HANCOCK, B. AND ZU ERMGASSEN, P.S.E. (eds) 2020. European Native Oyster Habitat Restoration Handbook. The Zoological Society of London, UK., London, UK

SHARMA, S., GOFF, J., MOODY, R.M., BYRON, D., HECK JR, K.L., POWERS, S.P., FERRARO, C. AND CEBRIAN, J., 2016. Do restored oyster reefs benefit seagrasses? An experimental study in the Northern Gulf of Mexico. Restoration Ecology, 24(3), pp.306-313.

SHELAMOFF, V., LAYTON, C., TATSUMI, M., CAMERON, M.J., WRIGHT, J.T. AND JOHNSON, C.R., 2019. Ecosystem engineering by a canopy-forming kelp facilitates the recruitment of native oysters. Restoration Ecology, 27(6), pp.1442-1451. Walne 1974

SPÄRCK, R. 1951. Fluctuations in the stock of oyster (Ostrea edulis) in the Limfjord in recent time. Rapports et Procès-verbaux des Réunions. Conseil Permanent International pour L'exploration de la Mer, 128, 27-29.

WOOLMER, A.P., SYVRET, M. AND FITZGERALD, A. 2011. Restoration of Native Oyster, Ostrea edulis, in South Wales: Options and Approaches. CCW Contract Science Report, no: 960, pp. 93.

ZU ERMGASSEN, P.S.E., GAMBLE, C., DEBNEY, A., COLSOUL, B., FABRA, M., SANDERSON, W.G., STRAND, Å. AND PRESTON, J. (eds) 2020. European Guidelines on Biosecurity in Native Oyster Restoration. The Zoological Society of London, UK., London, UK

Appendix 9 Seagrass beds

Seagrass restoration has been conducted worldwide for over 50 years but seagrass restoration efforts in the UK has been limited with restoration efforts limited to small-scale projects and trials. there are Two major techniques have been used:

- replanting;
    - transplantation of seagrass from donor populations
    - transplantation of cultured seagrass
    - transplanting seagrass cores or plugs, and
- re-seeding.

The most commonly used technique for seagrasses, is transplantation of seagrass from donor populations. Adult shoot replanting normally involves harvesting plants from an existing meadow and transplanting them to the restoration site. Transplanted material can include seedlings, sprigs, shoots, or rhizomes (van Katwijk and others, 2016). In most cases, some means of anchoring the shoots to the bottom is necessary until the roots can take hold (root into the bottom). In particular, van Katwijk and others (2016) pointed out that rhizome fragments, anchored using weights, were the most successful way to restore seagrass beds.

Replanting uses either labour-intensive diving techniques or various mechanistic approaches to plant various sizes and ages of seagrass plants into new localities.

The main UK seagrass species *Zostera marina* produces large amounts of seeds providing a potential simple, low cost opportunity for re-seeding. This approach has been utilised worldwide and may be amongst the most effective methods of restoration (van Katwijk and others, 2016). Projects planting seeds have used a range of methods such as spreading by hand at the water surface, the use of seed buoys (Pickerell and others, 2005), and more recently planting of seeds in coconut matting (Sousa and others, 2017). In many parts of the UK, the very large tidal ranges and resultant fast tidal currents can rapidly move seeds away from their intended location so that any technique that involves the loose spreading of seed (for example, seed buoys) is likely to be ineffective.

Unsworth and others (2019) trialled the use of hessian “Bags of Seagrass Seeds Line (BoSSLine)” for deploying seeds of the seagrass *Z. marina* over large scales at range of sites of high tidal range in West Wales adapted from previous studies. Two pilot studies were conducted, one at Porthdinllaen in North Wales and the other in the Helford River, Cornwall using differing types of seed bags under various deployment methods. A further seagrass seed bag experiment was then conducted at a further three sites around Wales. The locations were based on the use of a simple habitat suitability model and inspected using dropdown video and hand grabs to confirm suitability of the sediment. The sites were all in the range of 1–3 m depth (below low water spring) with a maximum tidal range of 7.68 m. Sediment type varied from fine and very fine sand at Dale and Longoar, to course sand at Freshwater East and all sites are fully marine. The average number of seeds in any given bag was 100 and the average (±SD) shoot density was 3.6 ± 2.1 and so we estimate that seed success was 3.6%. Excluding data from Freshwater East (where bags were smothered by sediment) we conclude that seed bags had a 94% success rate. The hessian bag not only keeps the seeds from dispersing due to tidal movements but also protects the seeds from burial or consumption. This technique has been adopted by other UK projects including the ReMEDIES project (see below).

Restoration using seeds and replanting techniques have sometimes been used together. Using seeds possibly in conjunction with adult plants, may in some instances prove more effective (van Katwijk and others, 2016).

In the UK, re-seeding using wild collected seed is being trialled at larger spatial scales. The LIFE Recreation ReMEDIES project, led by Natural England, will protect seagrass meadows and restore seagrass beds through training nearly 2,000 recreational users, to collect seed and replant seagrass, coupled with measures to protect seagrass by working with the recreational boating community to reduce the impact of recreational activities such as mooring and anchoring, utilising innovative technology such as Advanced Mooring Systems (AMS), best practice management techniques like voluntary codes, targeted training and behavioural change, and managing access. The four-year project aims to plant a total of eight hectares of seagrass meadows within five SAC areas which are currently in an unfavourable condition. In doing so, the project will demonstrate new habitat restoration and management approaches to seagrass restoration.

EUNIS Habitats recovery option is applicable to

A5.53 Sublittoral seagrass beds; A5.545 Zostera beds in reduced salinity infralittoral sediments

Seagrass beds: Natural recovery summary for applicable habitats

The recovery information presented here is summarised from the MarLIN assessment of A5.5331 *Zostera marina*/*angustifolia* beds on lower shore or infralittoral clean or muddy sand (D'Avack and others, 2019). That review and others on the MarLIN website provide more detail.

Zostera sp. and seagrasses are flowering plants adapted to an aquatic environment. They reproduce sexually via pollination of flowers and resultant sexual seed but can also reproduce and colonize sediment asexually via rhizomes. Seagrass species disperse and recruit to existing and new areas via pollen, seed, floating fragments or reproductive structures, vegetative growth (via rhizomes), and via biotic vectors such as wildfowl (e.g. geese). Boese and others (2009) found that natural seedling production was not of significance in the recovery of seagrass beds but that recovery was due exclusively to rhizome growth from adjacent perennial beds. However, genetic analysis of populations has revealed that sexual reproduction and seed are more important for recruitment and the persistence of seagrass beds than previously thought (Kendrick and others, 2012; 2017). Kendrick and others (2012; 2017) concluded that seagrass species are capable of extensive long distance dispersal based on the high level of genetic diversity and connectivity observed in natural populations.

Phillips and Menez (1988) state that seedling mortality is extremely high. Fishman and Orth (1996) report that 96% of Zostera marina seeds were lost from uncaged test areas due to transport (dispersal) or predation. Phillips and Menez (1988) note that seedlings rarely occur within the eelgrass beds except in areas cleared by storms, blow-out or excessive herbivory.

Seagrass reproduces vegetatively, i.e. by the growth of rhizome. Vegetative reproduction was thought to exceed seedling recruitment except in areas of sediment disturbance (Reusch and others 1998; Phillips and Menez 1988), although genetic analysis suggests a more complex process (Kendrick and others, 2012; 2017). *Zostera marina* plants are monomorphic, restricted to the horizontal growth of roots and, hence, unable to grow rhizomes vertically. This restriction to horizontal elongation of the roots makes the recolonization of adjacent bare patches difficult and explains why large beds are only found in gently sloping locations. A depression of the seabed caused by disturbance of the sediment can thus restrict the expansion of the bed. Larger denuded areas are likely to take longer to recover than smaller scars, for example, seagrass beds are likely to be more resilient to physical damage resulting from narrow furrows left after anchoring because of large edge to area ration and related availability of plants for recolonization. Manley and others (2015) reported a rhizome growth rate of 26 cm/yr. in *Zostera marina*.

Recruitment and recovery of seagrass meadows depend on numerous factors and is an interplay between seed recruitment to open or disturbed areas, the seed bank, and expansion by vegetative growth. Recruitment is also affected by local environmental conditions, and isolation due to coastal geomorphology such as islands and inlets, hydrography and even biological structures.

Reynolds and others (2013), estimated that natural recovery of *Zostera marina* seagrass beds in the isolated coastal bays of the Virginian coast, USA would have taken between 125 and 185 years to recover from the substantial decline due to wasting disease in the 1930s. Although small patches were observed in the 1990s seagrass was locally extinct for 60 years.

Seagrass beds: Overview of approach advantages when compared to natural recovery:

The UK has lost most of its seagrass. Of Britain's 155 estuaries, only 20 now contain seagrass: an 85% decline since the 1920s (Hiscock and others, 2005), with little natural recovery, and continuing losses and degradation in many parts of the country still being observed (Jones and Unsworth, 2016).

In many sites where historically seagrass beds may have occurred, the low dispersal and negative feedbacks may prevent recovery. In these instances, assisted recovery approaches are the only option for recovery.

It should be clear that where there are small losses of seagrass for example within mooring or anchoring scars assisted recovery or recession of bed fringes is not a cost-effective approach as the recovery timescale would probably be the same or similar. Instead, this approach is required to address historical losses and to restore ecosystem services and benefits.

Seagrass beds: Overview of approach disadvantages compared to natural recovery.

The range of techniques that have been used for restoration complicates the development of restoration trajectories for SAV beds. For example, seeding can result in different outcomes from transplanting (van Katwijk and others 2016) and the timing of planting, interannual variability in environmental conditions, use of fertilization, and genetics of the donor stock can also influence outcomes.

Seagrass restoration: economic costs

Moksnes and others (2021) provides an assessment of the economic cost of seagrass restoration including site selection and evaluation, harvesting, planting and monitoring. The 10-year cost per hectare is estimated to be SEK 1.2-2.5 million (£105, 000 – £220 000) for the shoot method and SEK 2.5-7.2 million (£218 692 - £633 065) for the seed method. The higher cost of the seed method is largely due to the labour required for seed collection (cited from Kent and others, 2021).

For the first 2 ha restoration project in Dale, Wales, the cost per hectare was approximately £200,000 per year (excluding monitoring costs) (costs cited from Kent and others, 2021).

Seagrass restoration benefits

Seagrass beds provide a range of ecosystem services and goods and benefits.

Education and engagement: Seagrass restoration projects have provided an opportunity for citizen science and engagement with the public (Jones and others, 2018).

Fisheries benefits, including the enhancement of fish nursery grounds. UK beds have been found to support commercially important juvenile fish such as plaice, pollock and herring relative to adjacent sand habitats (Bertelli and Unsworth, 2014).

Carbon sequestration: Carbon storage capacity of Zostera marina and Z. noltii beds were assessed by Potouroglou and others (2021) across 10 estuaries in Scotland. The organic carbon of vegetated (seagrass) areas was significantly higher than unvegetated areas (adjacent bare sand or mud), although variation was high.

Coastal erosion protection: seagrass beds is sediment stabilisation and shoreline protection. The presence of the grass absorbs wave energy and creates turbulence which causes sediment to fall out of suspension while the root system stabilises the seabed sediment, reducing resuspension of sediments and erosion (McGlathery and others, 2012; Moksnes and others, 2021).

Cultural benefits: as seagrass beds are found in shallow habitats often in sheltered bays, they are relatively accessible to the general public. This makes seagrass restoration projects a key candidate for community involvement, citizen science projects, community monitoring and a way for coastal groups to connect with the habitats on their doorsteps (Kent and others, 2021).

Seagrass beds: feasibility

Sixty one restoration sites have been identified in England, where restoration is considered feasible (MMO, 2019). Useful guides to seagrass restoration in the UK are the Seagrass Restoration handbook (Gamble and others, 2021) and the Seagrass restoration in Scotland - handbook and guidance (Kent and others, 2021).

Examples of UK projects include: Sky Ocean Rescue, WWF and Swansea University planting over 750,000 seagrass seeds in Dale Bay in Pembrokeshire to restore 20,000m^2. The project uses seeds from existing meadows around the British Isles, with over a million seeds due to be planted in total. The process of restoring seagrass started with a team of volunteer snorkellers and divers collecting the seeds from existing meadows around the country. Trials of seed bags show that success is supported by bag size (not too difficult for divers to handle), type of hessian (use 100% natural fibres to allow breakdown) and anchoring or pegging bags to reduce storm losses (Unsworth and others, 2019). Seed bags should be carefully sited to prevent sediment smothering.

Failures in many projects historically have been the result of limited consideration of the habitat requirements for seagrass and the continued presence of the stressor that caused the original seagrass loss (van Katwijk and others, 2016). A recent review of the success of restoration projects globally found that success relates to the severity 16 of the habitat degradation (eutrophication being worse than the combined impacts of dredging and filling or construction). The review also highlights the need for restoration to occur at sufficient scales in order to facilitate positive feedbacks and to spread the chances of success (van Katwijk and others, 2016). Large-scale planting increases plant survival. Scaling up spreads the risks, and the resultant increase in population growth rate enhances positive feedbacks, helping the seagrass to self-facilitate a more affable environment (van Katwijk and others, 2016). The positive effect of restoration scale on both trial survival and population growth rate of trials that survived suggests the existence of a threshold of scale of the trial required for restoration progress between 1000 and 10, 000 shoots/seeds. As the majority of seagrass restoration trials have been very small (55% analysed by van Katwijk and others, 2016 had fewer than 1000 specimens initially planted), this may explain the low trial survival rates recorded. A short distance to the donor site is also related to success. Whereas transplantations (replanting) frequently fail (60%) or have limited success, a substantial number of transplantations show large expansion rates as well (van Katwijk and others, 2016).

It is considered that some on-going management or inputs are likely to be required, such as gap-filling although no discussion was found in the evidence reviewed.

Seagrass beds: risks, challenges and uncertainties

Challenges: The presence of negative feedbacks that undermine natural recovery and restoration efforts have been identified (Maxwell and others, 2017). Poor water quality, is one of the largest threats faced by seagrass, both in the UK and globally (Jones and others, 2018). Loss of beds can create feedbacks with water flow velocity that can increase turbidity and hinder re-establishment, potentially explaining high failure rates for restoration attempts (van der Heide and others 2007).

Careful selection of a donor population is required both in terms of minimising the impact of collection on natural beds but also the chances of success. Research shows that when transplanting adult plants, it is important to match the conditions of the donor site and the restoration site (Moksnes and others, 2021). This is also important if using a seed-based approach although there is a greater potential for the plants to adapt to the new conditions as they grow. The genetic aspect of donor site seed collection and restoration site selection should also be considered with respect to the chances of success but also the impact on natural genetic diversity (Kent and others, 2021, Jahnke and others, 2015).

In summary, there is limited experience of seagrass restoration in the UK. Elsewhere, restoration/creation has been attempted, with very varying levels of success. van Katwijk et al, (2009) describe a series of guiding principles laid out by the Wadden Sea restoration project in order to maximise success rates:

1. Ensure long-term survival by promoting self-facilitation through implementation at a large-enough scale (hectares);
2. Focus on facilitating natural recovery through alleviating recruitment limitation (‘let nature work for you’);
3. Spread risks through space and time by restoring multiple sites on multiple occasions;
4. Keep the costs of restoration (per hectare) as low as possible to achieve an as-large-as-possible scale of success; and
5. Minimize impacts on source meadows while avoiding introductions of invasive species at restoration sites.

Seagrass beds: References

BERTELLI, C.M. AND UNSWORTH, R.K. 2014. Protecting the hand that feeds us: Seagrass (Zostera marina) serves as commercial juvenile fish habitat. Marine pollution bulletin, 83(2), pp.425-429.

BOESE, B.L., KALDY, J.E., CLINTON, P.J., ELDRIDGE, P.M. AND FOLGER, C.L. 2009. Recolonization of intertidal Zostera marina L. (eelgrass) following experimental shoot removal. Journal of Experimental Marine Biology and Ecology, 374 (1), 69-77.

D'AVACK, E.A.S., TYLER-WALTERS, H., WILDING, C. AND GARRARD, S.M., 2019. Zostera (Zostera) marina beds on lower shore or infralittoral clean or muddy sand. In Tyler-Walters H. and Hiscock K. (eds) Marine Life Information Network: Biology and Sensitivity Key Information Reviews, [on-line]. Plymouth: Marine Biological Association of the United Kingdom. [cited 14-03-2022]. Available from: https://www.marlin.ac.uk/habitat/detail/257

FISHMAN, J.R. AND ORTH, R.J. 1996. Effects of predation on Zostera marina L. seed abundance. Journal of Experimental Marine Biology and Ecology, 198, 11-26.

GAMBLE C., DEBNEY, A., GLOVER, A., BERTELLI, C., GREEN, B., HENDY, I., LILLEY, R., NUUTTILA, H., POTOUROGLOU, M., RAGAZZOLA, F., UNSWORTH, R. AND PRESTON, J, (EDS) 2021. Seagrass Restoration Handbook. Zoological Society of London, UK., London, UK.

HISCOCK, K., SELWELL, J., AND OAKLEY, J. 2005. The Marine Health Check 2005: A Report to Gauge the Health of the UK's sea life. Godalming: WWF-UK.

JAHNKE, M., I.A. SERRA, G. BERNARD, AND G. PROCACCINI. 2015. The importance of genetic make-up in seagrass restoration: A case study of the seagrass Zostera noltei. Marine Ecology Progress Series 532: 111–122. https://doi.org/10.3354/meps11355.

JONES, B.L. AND UNSWORTH, R.K., 2016. The perilous state of seagrass in the British Isles. Royal Society Open Science, 3(1), p.150596.

JONES, B.L., UNSWORTH, R.K., MCKENZIE, L.J., YOSHIDA, R.L. AND CULLEN-UNSWORTH, L.C., 2018. Crowdsourcing conservation: The role of citizen science in securing a future for seagrass. Marine pollution bulletin, 134, pp.210-215.

KENDRICK, G.A., ORTH, R.J., STATTON, J., HOVEY, R., RUIZ MONTOYA, L., LOWE, R.J., KRAUSS, S.L. AND SINCLAIR, E.A., 2017. Demographic and genetic connectivity: the role and consequences of reproduction, dispersal and recruitment in seagrasses. Biological Reviews, 92 (2), 921-938. DOI https://doi.org/10.1111/brv.12261

KENDRICK, G.A., WAYCOTT, M., CARRUTHERS, T.J.B., CAMBRIDGE, M.L., HOVEY, R., KRAUSS, S.L., LAVERY, P.S., LES, D.H., LOWE, R.J., VIDAL, O.M.I., OOI, J.L.S., ORTH, R.J., RIVERS, D.O., RUIZ-MONTOYA, L., SINCLAIR, E.A., STATTON, J., VAN DIJK, J.K. AND VERDUIN, J.J., 2012. The central role of dispersal in the maintenance and persistence of seagrass populations. BioScience, 62 (1), 56-65. DOI https://doi.org/10.1525/bio.2012.62.1.10

KENT, F., LILLEY, R., UNSWORTH, R., CUNNINGHAM, S., BEGG, T., BOULCOTT, P., JEORRETT, C., HORSBURGH, R. AND MICHELOTTI, M. 2021 Seagrass restoration in Scotland - handbook and guidance. NatureScot Research Report 1286.

MANLEY, S.R., ORTH, R.J. AND RUIZ-MONTOYA, L. 2015. Roles of dispersal and predation in determining seedling recruitment patterns in a foundational marine angiosperm. Marine Ecology Progress Series, 533, 109-120. DOI https://doi.org/10.3354/meps11363

MAXWELL, P.S., EKLÖF, J.S., VAN KATWIJK, M.M., O'BRIEN, K.R., DE LA TORRE-CASTRO, M., BOSTRÖM, C., BOUMA, T.J., KRAUSE-JENSEN, D., UNSWORTH, R.K., VAN TUSSENBROEK, B.I. AND VAN DER HEIDE, T., 2017. The fundamental role of ecological feedback mechanisms for the adaptive management of seagrass ecosystems–a review. Biological Reviews, 92(3), pp.1521-1538.

MMO 2019. Identifying sites suitable for marine habitat restoration or creation. A report produced for the Marine Management Organisation by ABPmer and AER, MMO Project No: 1135, February 2019, 93pp

MOKSNES, P. O., GIPPERTH, L., ERIANDER, L., LAAS, K., COLE, S., INFANTES, E. 2021. Handbook for restoration of eelgrass in Sweden - National guideline. Swedish Agency for Marine and Water Management, Report number 2021:5

PHILLIPS, R.C., AND MENEZ, E.G. 1988. Seagrasses. Smithsonian Contributions to the Marine Sciences, no. 34.

PICKERELL, C. H., SCHOTT, S., AND WYLLIE-ECHEVERRIA, S. 2005. Buoy-deployed seeding: Demonstration of a new eelgrass (Zostera marina L.) planting method. Ecol. Eng. 25, 127–136. doi: 10.1016/j.ecoleng.2005.03.005

POTOUROGLOU, M., WHITLOCK, D., MILATOVIC, L., MACKINNON, G., KENNEDY, H., DIELE, K. AND HUXHAM, M., 2021. The sediment carbon stocks of intertidal seagrass meadows in Scotland. Estuarine, Coastal and Shelf Science, 258, p.107442.

REUSCH, T.B.H., STAM, W.T., AND OLSEN, J.C. 1998. Size and estimated age of genets in eelgrass, Zostera marina, assessed with microsatellite markers. Marine Biology, 133, 519-525.

REYNOLDS, L.K., WAYCOTT, M., MCGLATHERY, K.J. AND ORTH, R.J., 2016. Ecosystem services returned through seagrass restoration. Restoration Ecology, 24(5), pp.583-588.

REYNOLDS, L.K., WAYCOTT, M. AND MCGLATHERY, K.J., 2013. Restoration recovers population structure and landscape genetic connectivity in a dispersal-limited ecosystem. Journal of Ecology, 101, 1288-1297. DOI https://doi.org/10.1111/1365-2745.12116

SOUSA, A.I., VALDEMARSEN, T., LILLEBØ, A.I., JØRGENSEN, L. AND FLINDT, M.R., 2017. A new marine measure enhancing *Zostera marina* seed germination and seedling survival. Ecological Engineering, 104, pp.131-140.

UNSWORTH, R.K., BERTELLI, C.M., CULLEN-UNSWORTH, L.C., ESTEBAN, N., JONES, B.L., LILLEY, R., LOWE, C., NUUTTILA, H.K. AND REES, S.C., 2019. Sowing the seeds of seagrass recovery using hessian bags. Frontiers in Ecology and Evolution, p.311.

VAN KATWIJK, M.M., THORHAUG, A., MARBÀ, N., ORTH, R.J., DUARTE, C.M., KENDRICK, G.A., ALTHUIZEN, I.H., BALESTRI, E., BERNARD, G., CAMBRIDGE, M.L. AND CUNHA, A., 2016. Global analysis of seagrass restoration: the importance of large-scale planting. Journal of Applied Ecology, 53(2), pp.567-578.

VAN KATWIJK, M.M., BOS, A.R., DE JONGE, V.N., HANSSEN, L.S.A.M., HERMUS, D.C.R. AND DE JONG, D.J., 2009. Guidelines for seagrass restoration: importance of habitat selection and donor population, spreading of risks, and ecosystem engineering effects. Marine pollution bulletin, 58(2), pp.179-188.

VAN DER HEIDE, T., VAN NES, E.H., GEERLING, G.W., SMOLDERS, A.J., BOUMA, T.J. AND VAN KATWIJK, M.M., 2007. Positive feedbacks in seagrass ecosystems: implications for success in conservation and restoration. *Ecosystems*, *10*(8), pp.1311-1322.

Appendix 10 Kelp and seaweed restoration

Assisted recovery of kelp and seaweed habitats has been conducted internationally for over 60 years with increased interest and research within the last couple decades. To date approaches trialled in the UK have been small-scale. Assisted recovery timescales are considered to be similar to natural recovery potential to allow canopy forming species to reach a comparable size and age structure to unimpacted habitats. Most studies focus on the recovery of ecosystem engineers especially kelp. There has been no kelp or seaweed restoration experiments in the UK (Earp and others, 2022). Despite this, many assisted recovery studies involved species commonly found around the UK, particularly Laminarian species (Earp and others, 2022). Kelp and seaweed assisted recovery techniques are summarised below.

Kelp and seaweed transplantation

Transplantation is the most common technique for Laminarian and Fucalean species (Earp and others. 2022) which are common around the UK coastline. It involves collecting juvenile or adult individuals from existing populations (Eger and others, 2021). Transplanting older life-history stages is more successful as they are less susceptible to grazing and competitive exclusion (Morris and others, 2020). The transplant holdfasts are then attached to an artificial material placed on the sea floor. The materials used include small concrete blocks (Oyamada and others, 2008), ropes (North, 1978) and mesh mats anchored to the sea floor (Campbell and others. 2014). This process makes transplantation a labour intensive and logistically complex technique (Eger and others, 2021) The aim is for the holdfast to transfer its attachment to the benthos (which occurs if the holdfast isn't too damaged) and act as a source of recruits to create a new population (Morris and others, 2020). For some species, transplants are only successful in the presence of adult kelps nearby to the area being restored (Layton and others, 2019; Eger and others, 2021).

Seeding

Assisting recovery through seeding involves the collection of fertile kelp material or the culturing of sporophytes from an existing kelp population. Weighted mesh bags are filled with the fertile material and placed in recovery area (Eger and others, 2021; Choi and others, 2000). Attaching the bags to the seabed can improve the likelihood of recruitment to the desired area (Morris and others, 2020). Earlier kelp and seaweed life-stages experience the greatest mortality rates so seeding requires large amounts of fertile material to be successful (Morris and others, 2020). Seeding for kelp and seaweed restoration has produced limited success (Eger and others, 2021) and as a result isn't widely used. Earp and others (2022) showed for Fucalean and Laminarian species seeding was less successful than transplanting, however it was acknowledged that seeding is relatively understudied. Seeding is a labour and time intensive technique however it has potential to be a larger scale, more cost-effective method if some of the challenges can be overcome (Saunders and others, 2020; Eger and others, 2021).

Green Gravel

A newly developed method (Fredriksen and others, 2020) where small stones are seeded with kelp and cultured under optimal laboratory conditions prior to out-planting or deployment at sea. The method has been successfully trialled with Sugar Kelp *Saccharina latissima* in Norway, and is potentially highly scalable as, although lab costs are high, gravel can be deployed from boats avoiding labour intensive and costly SCUBA diver installation methods. Boat based deployment also allows potentially inaccessible underwater habitats to be restored. Smaller substrates are also easier to handle and transport than mature transplants or artificial substrates, and the technique appears to be fairly robust (Fredriksen and others, 2020). Research is currently underway to explore limitations of the technique, such as its applicability to sites with high wave energy or currents. The Green Gravel Action Group (www.greengravel.org) aims to test this novel and cost-effective technique across an array of environmental contexts and species.

Removal of grazers

UK Kelp and seaweed species experiences relatively little grazing pressure from Urchins (Burrows and others, 2014) and therefore the removal of grazers offers little potential to assisted recovery in the UK.

Removal of competitors

Fast-growing, turf macroalgae species can competitively exclude habitat-forming species. Their removal has been utilised in Japan to assist kelp recovery. Rock is cleared of “low-value” primary producers through mechanical and manual removal (Eger and others, 2021). Removal of competitors has high restoration success rates but is usually used in combination with other strategies such as transplanting. Studies that solely use removal of competitors are few and rely on proximity to nearby populations to provide recruits of the species targeted for restoration (Earp and others, 2022).

Artificial reefs

This strategy installs man-made structures onto the sea floor. The material used in the structures can range from rocks to sunken ships (Eger and others, 2021; Tickell and others, 2019) and are designed to stimulate algal recruitment and growth. Artificial reefs are ecologically engineered through material choice and shape to increase structural complexity and therefore the settlement area for algae on the artificial reef (Morris and others, 2020). Often artificial reefs are used in combination with transplantation and seeding to provide a source of propagules. It is easier to attach transplants and seeding bags to artificial reefs than the sea floor making them beneficial to restoration projects that utilise multiple techniques (Eger and others, 2021). Some designs infuse reef materials with iron and nitrates that are slowly excreted over time and encourage growth in colonising algae (Oyamada and others, 2008).

EUNIS Habitats recovery option is applicable to

The approaches are relevant to kelp dominated habitats:

- A3.11 Kelp with cushion fauna and/or foliose red seaweeds,
- *A3.12 Sediment-affected or disturbed kelp and seaweed communities
- A3.21 Kelp and red seaweeds (moderate energy infralittoral rock)
- *A3.22 Kelp and seaweed communities in tide-swept sheltered conditions
- *A3.31 Silted kelp on low energy infralittoral rock with full salinity
- *A3.32 Kelp in variable salinity on low energy infralittoral rock

*Feasibility is likely to be lower due to the disturbed conditions/currents

Natural recovery summary for applicable habitats:

This Natural recovery summary focusses on habitat forming UK macroalgal species.

Laminaria hyperborea is the dominant canopy-former on most sub-littoral reefs around the UK and Ireland. The natural recovery rate of L. hyperborean beds is slower than similar species at around 2-6 years (Kain, 1979; Birkett and others, 1998; Christie and others, 1998). The recovery of associated epiphytic communities takes around 10 years based on data from discrete kelp harvesting events. Zoospore dispersal is greatly influenced by water movements and the rate of successful fertilization decreases as they disperse further from their parental source. The dispersal range is estimated at around ~200m (Fredriksen and others, 1995). Consequently, recovery rates in disturbed areas are influenced by the proximity to existing *L. hyperborea* beds that can provide recruits (Kain, 1979; Fredriksen and others, 1995). Competitive interactions with *Undaria pinnatifida* and *Laminaria ochraleuca* can influence *L. hyperborea* recovery rates (Smale and others, 2013, Smale and others, 2015, Brodie and others, 2014). Predicted sea temperature rises in the North and Celtic seas (Philippart and others, 2011) will favour these species and prevent the recovery of *L .hyperborea* populations in disturbed areas. The limited Urchin grazing in the UK lends the habitat a medium level of resilience with natural recovery rates longer than other macroalgal species but considerably shorter than some other coastal habitats.

Natural recovery of *Laminaria digitata* occurred within 18-24 months post harvesting (Kain 1979, Engelen and others, 2011, Smith, 1985). Despite the relatively quick recolonization of these areas the return of the biotope to its original condition will lag. There are seasonal differences in the recovery rates with autumn recovery being more rapid. The dispersal and resulting recruitment of *L. digitata* zoospores has been recorded 600m from the parental plants however vary with local water movement (Brennan *and others,* 2014). As a result, recovery is slower in areas isolated from other *L. digitata* populations. Opportunistic, competitive species can out-compete and prevent the recovery of *L. digitata* populations if frequency disturbance is high.

Alaria esculenta is an opportunistic colonizing species found in the North Atlantic and dominates areas exposed to severe wave action. They exhibit high growth rates which can exceed 20cm/month during April-May (Birkett and others, 1998). *A. esculenta* appears early in algal succession and becomes the dominant algae within 9 months suggesting fast recovery rates. However, the low zoospore dispersal capacity, reported to be within 10m, means nearby parental sources are required to facilitate natural recovery (Sundene, 1962; Norton 1992). Despite this the resilience of this species is high and the natural recovery can be rapid.

Saccharina latissima and *Saccorhiza polyschides* are fast-growing opportunistic seaweeds that colonise early after disturbances. Leinaas and Christie (1996) found that *S. latissima* colonised disturbed areas 2 weeks after disturbance events. *S. latissima* and *S. polyschides* reach maturity comparatively quickly at around 17 months and 8 months respectively (Birkett and others, 1998). *S.latissima* grows faster than other perennial kelp species exhibiting their highest growth rates after recruitment in the late winter to early spring (Lüning, 1990; Birkett and others, 1998). Spores are subsequently released from autumn to winter. *S.polyschides* recruitment takes place in March-April and rapid growth occurs from this point until maturity is reached which triggers a phase of senescence where growth ceases (Birkett and others, 1998). Their zoospores have large dispersal ranges (Fredriksen and others, 1995). As a result of the colonization ability and growth rates of *S.latissima* and *S.polyschides*, they are considered resilient species whose populations can naturally recover rapidly.

Recovery potential of these species and their created habitats relies upon the abiotic conditions of their location. The frequency and intensity of disturbances will also impact recovery rates with more frequent and intense disturbances lengthening recovery times (Burrows and others, 2014;

Birkett and others, 1998). Most species rely upon nearby populations to provide recruits that can recolonize disturbed areas.

Overview of approach advantages when compared to natural recovery

For Laminarian kelp forests, significant declines have occurred in 38% of the ecoregions where long-term data is available. Kelp and habitat-forming seaweed population declines coupled with low dispersal potential (Earp and others, 2022*)* and the importance of the presence of adult conspecifics for juvenile growth (Eger and others, 2020a) could prevent natural recovery in certain areas. In these areas assisted recovery approaches that provide a source of recruits are the only strategies to recovering the habitat *(*Earp and others, 2022).

Climate change represents a risk to the UK and Irelands macroalgal assemblage structure and ecosystem function. Warming North and Celtic sea temperatures is thought to produce range expansions further North in more southerly-distributed species such as *L. ochroleuca* and *S. polyschides* (Burrows and others, 2014). If some disturbed areas are left to naturally recover these southerly-distributed species could potentially outcompete more northerly-distributed species such as *L. digitata, L. hyperborea* and *A. esculenta* leading to shifts in associated assemblages and ecosystem functioning. Assisted recovery provides more control over the recovered habitat and could potentially be used to encourage *L. hyperborea* and *L. digitata* to re-colonise areas to maintain their associated habitats. Predicted increases in storminess, due to climate change, will likely damage and dislodge canopy forming macroalgae. This will alter patch dynamics and impact the natural recovery ability of kelp beds and associated environments (Burrows and others, 2014). Increased disturbances could produce negative feedback considering the low dispersal potential of kelp propagules (Earp and others, 2022; Burrows and others, 2014) and the importance of the presence of adult's conspecifics for juvenile growth (Eger and others, 2020a). Consequently, complex macroalgal habitats could experience state-shifts to simple algal turf dominated systems if disturbed areas (Burrows and others*,* 2014) are allowed to naturally recover. Once these alternate-state shifts occur it is unlikely the ecosystem will naturally return to its original state. Manual removal and transplantation or seeding could provide the disturbance and source of propagules to return these areas to the desired canopy-forming macroalgal habitat (Morris and others, 2020).

Assisted recovery provides the potential of a genetic baseline to be selected, defined as the level of genetic diversity and structure chosen and initially replicated in assisted recovery projects (Coleman and others, 2020). The potential to reinforce existing genetic baselines by increasing the genetic diversity of the recovered populations could provide the adaptive capacity to cope with projected increases in environmental and anthropogenic stress (Coleman and others, 2020). Redefining genetic baselines could address climate-driven loss of UK *L. hyperborea, L. digitata* and *A. esculenta* habitats. Donor material used for assisted recovery (transplants and seeding) could be selected based upon their genetics. Genotypes that are more resilient to higher temperatures could be identified through experimentation and selected for assisted recovery projects (Eger and others, 2020a, Coleman and others, 2020). Synthetic biology and gene editing could be a novel way to introduce beneficial genetics and engineer adaptation through assisted recovery programmes (Coleman and others, 2020). These techniques could mitigate climate change associated losses of certain macroalgae and manufacture resilience to future change.

Overview of approach disadvantages compared to natural recovery:

The transplantation process results in the removal of individuals from and disturbance to an existing population. Due to low survival and recruitment rates, many transplants, in some cases thousands (North, 1978), are needed for the successful recovery of a habitat (Morris and others, 2020). Acquiring these individuals, especially adults could affect the persistence of the extant donor population. Transplanting juveniles could reduce the impact on donor populations (Morris and others, 2020), however, due to the higher vulnerability of younger life-history stages, this would impact the success of the assisted recovery effort. The removal of reproductive material for seeding would also impact the extant population, although to a lesser extent (Morris and others, 2020). Assisted recovery could also potentially cause deleterious effects to a kelp population by introducing foreign alleles through transplanted or seeded individuals (Eger and others, 2020a; Filbee-Dexter and Smajdor, 2019). This could result in outbreeding depression and the loss of rare but locally adapted alleles, having a negative impact on the genetic resilience and therefore possibly the persistence of nearby natural population (Earp and others, 2022).

Installing artificial reefs will permanently change the habitats they are installed in. Compared to natural recovery, the ecosystem created will vary and it is difficult to predict the differences. This trade-off is a societal decision but could be seen as a disadvantage (Eger and others, 2021).

Transplanted individuals also usually have lower survival rates than those in natural populations, probably due to holdfast damage during the removal and reattachment process (Earp and others, 2022).

Kelp assisted recovery: economic costs

There is a lack of research on Kelp restoration costs of UK based projects. Most of the information used in this section is from international projects with geographical differences in biodiversity and environmental conditions to the UK which can lead to different restoration challenges. As a result, the information provided could be accurate indicators of the potential costs of Kelp restoration projects in the UK, but variation is expected. There was no evidence to assess or separate capital, operation and maintenance costs. In line with other methods, capital costs for feasibility studies, licensing and

The lowest cost restoration techniques aim to control sea urchin populations to reverse ecosystem state shifts from kelp forests to sea urchin barrens. There is little evidence for the occurrence of sea urchin barrens in the UK (Burrows and others, 2014), therefore, the use of these techniques in UK kelp restoration projects will be limited. Seeding, transplanting and artificial reefs will be the more likely used kelp restoration techniques. Seeding and transplanting have similar causes of costs including materials, transport and fuel, equipment and hourly rates for manual labour (e.g. divers and drivers). (Carney and others, 2005; Campbell and others, 2014). General restoration project requires project planning involving literature reviews, site selection, population monitoring and dealing with logistical constraints which also contribute to overall costs (Campbell and others, 2014; Groeneveld and others, 2019; Cebrian and others, 2021). Table 18 shows these techniques have considerably higher costs per hectare with an average of £285,742 and £367,910 for seeding and transplanting respectively.

The MERCES project included transplanting *L. hyperborea* and *S. latissima* in Northern Norway. They costed the project at £9,188 to transplant an area of 100m^2 (Groeneveld and others, 2019), more expensive than Eger and others (2021) estimates. This observed difference in costing estimates makes it difficult to accurately estimate the costs of these techniques. It should be noted the costs of non-consumable laboratory equipment (e.g. stereomicroscopes or diving materials) needed for these restoration projects are not included in estimates as pre-existing facilities are usually utilised. The distance between sites and the required facilities strongly affects the cost-effectiveness of the restoration efforts because transport conditions and duration are critical to the survival of recruits and germlings (Cebrian and others, 2021). An example of monitoring costs come from the Operation Crayweed project with an estimate of the monitoring and management costs for eleven restoration sites with 6, 2m^2 restored patches *of P. comosa* at around $18,500 per annum (Layton and others, 2020; Campbell and others, 2014).

Despite the greater cost of seeding than transplanting reported by Eger and others, (2020b) some projects have reported the opposite for *Cystoseira* spp. and *Nereocytis leutke*ana (Verdura and others, 2018; Carney and others, 2005).

Kelp restoration seems more costly compared to the restoration of other marine habitats. However, there has been relatively few kelp restoration projects and even less information is available on the costs they generate. It is reasonable to assume that as the kelp restoration field advances the techniques used will be refined, improving the efficiency and reducing the costs (Eger and others, 2021). Selecting sites connected to intact kelp populations and short distances from the required restoration facilities is also thought to potentially reduce project costs (Morris and others, 2020; Layton and others, 2020). Further research into the ecological processes that underpin the survival and proliferation of the targeted kelp species will help to guide project planning (including site selection) and improve techniques, therefore diminishing costs (Campbell and others, 2014).

Economies of scale should result in the larger projects costing less per hectare (Turner and Boyer, 1997). Morris and others, (2020) review of kelp restoration projects found 56% of the projects were small scale (restoration area of or under 100m2). Future upscaling of kelp restoration projects could potentially reduce the cost per hectare of the techniques mentioned (Eger and others, 2021). A potential example of this could be artificial reefs in Norway costing around £183,765 for an area of 500m^2 (Groeneveld and others, 2019), a higher price per hectare than the average £384,227 estimated by Eger and others, (2021). However, it is projected for the phase 3 expansion of the Wheeler North reef project which will result in 85 hectares of artificial reef, to cost between $17.62 - $27.89 million (USD, 2010; Southern California Edison, 2017), likely leading to a significantly lower cost per hectare than Eger and others, (2021) estimated average.

Fredriksen (2021) estimated the costs of the green gravel technique to be smaller than other active methods, at $7 USD m^2 ($70,000/ha). This was calculated based on the operational costs for staff time, materials and fuel needed to produce and deploy 116 kg green gravel. However, in keeping with other restoration projects, subsequent monitoring, bench fees, vessel, vehicles and other fixed infrastructure costs were not included. By comparison, they estimated costs for seeding at $48-118 m^2 ($480,000-$1,180,000/ha), transplanting at $6-160 m^2 ($60,000- $160,000/ha), herbivore removal at $2 m^2 (£20,000/ha) and artificial reefs at $8 USD m^2.

As an illustrative UK example, approximate costings are provided for pilot trials using the green gravel technique to seed *S. latissima* in Plymouth, which are ongoing in 2022 (Wilding, pers comm). Grow-out in the laboratory is estimated to take three months, prior to deployment at sea and monitoring over the course of the following year. Planning and preparation, including collection of fertile material, spore extraction, cleaning and preparation of gravel substrates, inoculation, laboratory cultivation (including cleaning, water changes, preparation of nutrients, and modification of lighting intensity) was estimated at five days for a team of two staff (costed at £300 per day), totalling £3,000. Deployment and monitoring of success will require 5 days of a commercially qualified dive team and vessel, costed at £1,000 per day. Consumables (nutrient media, lab equipment, filtered sea water) totalled £1,5000 and use of the aquarium facilities (tanks, chillers, air supply, lighting, nutrient media, technician time) at £2,000. Therefore the total is estimated at £11,500 to seed a small area (2m^2) at four sites.

Table 18 Provides summary costs for different techniques. For the costs cited from Elger and others (2021), values were converted to GBP using the 2010 average exchange rate and rounded to the nearest pound. Costs from Fredriksen and others (2021) were converted to GBP using the current rate (I.31 USD).

Table 18. Summary of the average costs of Kelp restoration methods, provided by an extensive literature review on the costings of kelp restoration methods from Eger and others, (2021).

Restoration Method	Aim	Average cost per hectare	Reference
Quickliming	Controlling Sea Urchins	~£842 ($1300 USD)	Eger and others, 2021
Manual Removal	Controlling Sea Urchins	~£28,361 ($43800 USD)	Eger and others, 2021
Seeding	Increase Kelp Recruitment	~£285,742 ($441300 USD)	Eger and others, 2021
Seeding	Increase Kelp Recruitment	£367,786- 904,141 ($480,000- $1,180,000)	Fredriksen and others, 2021
Transplanting	Increase Kelp Population	~£367,910 ($582100 USD)	Eger and others, 2021
Transplanting	Increase Kelp Population	£45973-£122,595 ($60,000-$160,000	Fredriksen and others, 2021
Transplanting	Increase Kelp Population	£918,800	(Groeneveld and others, 2019),
Artificial Reef	Increase Kelp Recruitment	~£384,227 ($593400 USD)	Eger and others, 2021
Green gravel	Transplant kelp	£53,635 ($582100 USD)	Fredriksen and others, 2021

Kelp and seaweed assisted recovery: Benefits

Large scale kelp restoration requires large financial inputs. Many governments will attempt to stimulate their economies by funding large infrastructure projects. Viewing kelp and seaweed restoration in this way illustrates the substantial socioeconomic benefits it could have (Eger and others, 2021) such as the creation of jobs from the labour required to enact and monitor projects.

Assisted recovery of kelp and seaweeds aims to restore their populations and the associated ecosystem and biodiversity. Therefore, the benefits of restoring kelp and seaweeds are equivalent to the ecosystem services that those habitats provide. A summary of the ecosystem services kelp provide in the UK is detailed below following the structure of a previous paper on this subject (Smale and others, 2013).

Biodiversity: UK kelp species are considered ecosystem engineers and play a central part in the habitat structure and ecological assemblages found in the nearshore area. Kelps also initiate habitat cascades where sessile flora and fauna that kelps provide habitat for, in-turn provide habitats for other marine organisms (Teagle and others, 2017). Within the UK over 1800 species have been recorded in kelp-dominated habitats. Out of the UK kelp species *L. hyperborea* fosters the greatest biodiversity in its associated assemblages which is thought to be due to differences in its morphology. Kelp undeniably supports a vast array of marine life (Smale and others, 2013; Steneck and others, 2002).

Productivity and food webs: Kelp forests are one of the most productive habitat types on earth and may account for around 45% of primary production in the coastal waters in the UK (Smale and others, 2013). Some kelp biomass is directly consumed by herbivorous organisms, however, more than 80% of kelp production enters nutrient cycles as detritus or dissolved organic matter (Krumhansl and Scheibling, 2012). Kelp detritus is kept within kelp forests or carried by water movements to adjacent habitats. It represents a source of energy to suspension feeders, detritovores and other consumers of organic material (Smale and others, 2013) and this source is especially important to low-productivity habitats such as sandy beach habitats (Ince and others, 2007).

Resource provision: Kelp has a myriad of uses. It can be processed and used as feed supplements in agriculture and aquaculture. Kelps is rich in nutrients and alginates so is collected and used as fertilizer. Alginates are extracted and used in food, textile and pharmaceutical industries (Smale and others, 2013). The European lobster, velvet swimming crabs and seasonal spider crabs (*Homarus gammarus*, *Necora puber* and *Maja brachydactyla* respectively) rely on kelp forests for habitat and prey. Kelp forests provide a habitat to commercially important fish species including Atlantic cod, European sea bass and Pollack (Holdbrooks and others, 1990; Steneck and others, 2002). The Kelp itself can also be consumed and some small-scale suppliers exist around the UK and Ireland. Kelp is a potential source of biofuel which could help to mitigate the causes of climate change. Globally marine macroalgae captures and sequesters significant amounts of atmospheric carbon dioxide helping to regulate climate. Recent research shows the considerable potential for macroalgae to be sequestered to deeper waters providing a long-term store of carbon (Bayley and others, 2021).

Socioeconomic importance: Macroalgae and the associated biodiversity holds great socioeconomic value. It's been calculated that coastal marine biodiversity contributes over £11 billion to the UK economy through recreational industries. The commercial industries associated with kelp and its biodiversity create jobs and contributes to the economy (Smale and others, 2013).

Kelp and seaweed assisted recovery: Coastal defence:

Kelp forests and seaweeds can prevent or reduce damage caused by storm surges through wave damping and attenuation. By changing the water motion (Eckman and others,1989) and reducing the velocity of breaking waves they reduce coastal erosion (Türker and others, 2006). This provides a degree of coastal defence that protects coastal communities and structures, which will become increasingly important because of predicted sea-level rise and increasing intensity and frequency of storms (Smale and others, 2013).

Kelp and seaweed assisted recovery: Cost-benefit analysis

Eger and others, (2021) valued the ecosystem services of 4 major kelp genera (including Laminaria) at around $135,200 to $177,100/ha/year. This value the habitat provides could potentially offset the costs of kelp restoration within 2-7 years.

Kelp and seaweed assisted recovery: feasibility

Eger and others, (2021) reported that disturbance events are regular causes of project failures. Urchin grazing is one of these disturbance events. The lack of this pressure could increase the feasibility of the assisted recovery of macroalgal species in the UK. Other disturbance events such as consistently warmer sea temperatures, marine heat waves and storms have caused transplants and individuals that recruited from seeding to die off. The feasibility of kelp and macroalgal community assisted recovery relies on mitigating these environmental barriers to success (Eger and others, 2021).

There are logistic challenges to assisting the recovery of kelp in subtidal environments that affect the feasibility projects. Equipment such as a boat, SCUBA equipment, artificial materials and more is needed for implementing assisted recovery techniques and for scientific monitoring (Layton and others, 2020).

Knowledge of the local conditions of the targeted area to recover is crucial to understanding the feasibility of a project. In some areas multiple stressors need to be addressed and require the combination of multiple assisted recovery strategies. The economic and time investments into these projects will be higher for them to be successful (Eger and others, 2021).

Research shows that kelp populations have density thresholds that alter the environment and support future generations. The failure to re-establish intraspecific facilitation though proximity to adult individuals of the same species could explain the failure of some kelp restoration efforts (Layton and others, 2019, 2020). Proximity of other kelp populations is a key predictor of project success (Eger and others, 2021).

Successful kelp restoration projects are rare and require large economic and time investments. (Bayrakatarov and others, 2016; Eger and others, 2020b Layton and others, 2020). With the development of the field this could potentially change (Eger *and others,* 2020b) however it is clear that financial support is crucial to making kelp restoration projects at relevant scales feasible (Eger and others, 2020b).

Kelp and seaweed assisted recovery: risks, challenges and uncertainties

Most of the work undertaken has been on small spatial scales in a specific area, considering one or a small number of species and often monitoring is limited in scope and duration (Eger and others, 2020a; Earp and others, 2022). The scalability of these techniques is uncertain and whether they are successful in a UK context has not been investigated. Restoration approaches might produce habitat differences compared to natural recovery over time (Earp and others, 2022). Also, the seasonal variations in growth, survival and dislodgment rates might mean that results of studies with limited monitoring times might lead to the misrepresentation of success rates (Earp and others, 2022, De La Fuente and others, 2019).

Restoring kelp and seaweeds in areas without populations nearby is a challenge (Eger and others, 2021). An uncertainty is the density and number of transplants needed to provide intraspecific facilitation that’s influential for restoration project success.

A lack of UK based kelp and seaweed restoration projects creates uncertainties around the application of techniques developed for different species and locations in the UK. Gleason and others, (2021) have produced guidelines to making approach and management decisions for kelp restoration in California. These key principles include: 1. Problem formulation which requires knowledge of the target species, their environmental and anthropogenic stressors, the spatial scale, stakeholder engagement, and logistical, financial and policy constraints; 2. Setting clear objectives involves what the project aims are and how it will achieve them; 3. Identifying alternatives includes identifying alternative actions that will meet the objectives defined; 4. Predicting consequences uses information of the pre-restored system to identify potential impacts caused by the restoration project; 5. Evaluating trade-offs of different restoration actions considering the objectives; 6. Making decisions based on the previous steps; 7. Act, monitor and learn by implementing techniques decided upon and monitor progress to allow for adaptive management.

Recommendations for future research/work required

Future work and research should provide information on the environmental conditions of the targeted restoration area to allow for comparisons of the factors influencing the successes and failures of assisted recovery strategies. Long-term monitoring and reporting of the state of recovered areas should be undertaken to determine whether the assisted recovery of macroalgal species on their own has the potential to restore the complex communities and their services associated with macroalgal dominated habitats (Earp and others, 2022).

Hypothesised strategies which would increase the success and feasibility whilst decreasing the costs could be investigated. These include using techniques from the aquaculture industry such as culturing ropes and suspending them in the ocean to act as a source population (Eger and others, 2021). Future work on how to mitigate the environmental barriers to assisted recovery project success is needed (Eger and others, 2021).

Kelp and seaweed assisted recovery: references

BAYLEY, D., BRICKLE, P., BREWIN, P., GOLDING, N. AND PELEMBE, T. 2021. Valuation of kelp forest ecosystem services in the Falkland Islands: A case study integrating blue carbon sequestration potential. One Ecosystem, 6, e62811.

BAYRAKTAROV, E., SAUNDERS, M.I., ABDULLAH, S., MILLS, M., BEHER, J., POSSINGHAM, H.P., MUMBY, P.J. AND LOVELOCK, C.E. 2016. The cost and feasibility of marine coastal restoration. Ecological Applications, 26(4), 1055-1074.

BIRKETT, D.A., MAGGS, C.A., DRING, M.J. AND BOADEN, P.J.S., 1998. Infralittoral reef biotopes with kelp species: an overview of dynamic and sensitivity characteristics for conservation management of marine SACs. Natura 2000 report prepared by Scottish Association of Marine Science (SAMS) for the UK Marine SACs Project., Scottish Association for Marine Science. (UK Marine SACs Project, vol VI.), 174 pp.

BRENNAN, G., KREGTING, L., BEATTY, G.E., COLE, C., ELSÄßER, B., SAVIDGE, G. AND PROVAN, J. 2014. Understanding macroalgal dispersal in a complex hydrodynamic environment: a combined population genetic and physical modelling approach. Journal of the Royal Society Interface, 11(95), p.20140197.

BRODIE, J., WILLIAMSON, C.J., SMALE, D.A., KAMENOS, N.A., MIESZKOWSKA, N., SANTOS, R., CUNLIFFE, M., STEINKE, M., YESSON, C., ANDERSON, K.M. AND ASNAGHI, V. 2014. The future of the northeast A tlantic benthic flora in a high CO 2 world. Ecology and Evolution, 4(13), pp.2787-2798.

BURROWS, M.T., SMALE, D., O'CONNOR, N., VAN REIN, H. AND MOORE, P. 2014. Marine strategy framework directive indicators for UK kelp habitats part 1: developing proposals for potential indicators.

CAMPBELL, A.H., MARZINELLI, E.M., VERGES, A., COLEMAN, M.A. AND STEINBERG, P.D. 2014. Towards restoration of missing underwater forests. PloS one, 9(1), p.e84106.

CARNEY, L.T., WAALAND, J.R., KLINGER, T. AND EWING, K... 2005. Restoration of the bull kelp *Nereocystis luetkeana* in nearshore rocky habitats. *Marine Ecology Progress Series*, 302, 49-61.

CEBRIAN, E., TAMBURELLO, L., VERDURA, J., GUARNIERI, G., MEDRANO, A., LINARES, C., HEREU, B., GARRABOU, J., CERRANO, C., GALOBART, C. AND FRASCHETTI, S. 2021. A Roadmap for the Restoration of Mediterranean Macroalgal Forests. Frontiers in Marine Science, 1456.

CHOI, C.G., SERISAWA, Y., OHNO, M. AND SOHN, C.H. 2000. Using the spore bag method. Algae, 15(3), pp.179-182.

CHRISTIE, H., FREDRIKSEN, S. AND RINDE, E. 1998. Regrowth of kelp and colonization of epiphyte and fauna community after kelp trawling at the coast of Norway. *Hydrobiologia*, 375/376, 49-58.

COLEMAN, M.A., WOOD, G., FILBEE-DEXTER, K., MINNE, A.J., GOOLD, H.D., VERGÉS, A., MARZINELLI, E.M., STEINBERG, P.D. AND WERNBERG, T. 2020. Restore or Redefine: Future Trajectories for Restoration. *Frontiers in Marine Science,* 7, 237.

DE LA FUENTE, G., CHIANTORE, M., ASNAGHI, V., KALEB, S. AND FALACE, A. 2019. First ex situ outplanting of the habitat-forming seaweed Cystoseira amentacea var. stricta from a restoration perspective. *PeerJ*, 7, e7290.

ECKMAN, J.E., DUGGINS, D.O. AND SEWELL, A.T. 1989. Ecology of under story kelp environments. I. Effects of kelps on flow and particle transport near the bottom. *Journal of Experimental Marine Biology and Ecology*, 129(2), 173-187.

EGER, A., MARZINELLI, E., CHRISTIE, H., FUJITA, D., HONG, S., KIM, J.H., LIEE, L.C., MCHUGH, T.A., NISHIHARA, G.N., VASQUEZ, A.P.G. AND TATSUMI, M. 2021. Global Kelp Forest Restoration: Past lessons, status, and future goals. https://doi.org/10.32942/osf.io/emaz2

EGER, A.M., MARZINELLI, E., GRIBBEN, P., JOHNSON, C.R., LAYTON, C., STEINBERG, P.D., WOOD, G., SILLIMAN, B.R. AND VERGÉS, A. 2020a. Playing to the Positives: Using Synergies to Enhance Kelp Forest Restoration. *Frontiers in Marine Science,* 7, 544.

EGER, A.M., VERGÉS, A., CHOI, C.G., CHRISTIE, H., COLEMAN, M.A., FAGERLI, C.W., FUJITA, D., HASEGAWA, M., KIM, J.H., MAYER-PINTO, M. AND REED, D.C. 2020b. Financial and Institutional Support Are Important for Large-Scale Kelp Forest Restoration. *Frontiers in Marine Science*, 7, 535277.

ENGELEN, A.H., LEVEQUE, L., DESTOMBE, C. AND VALERO, M. 2011. Spatial and temporal patterns of recovery of low intertidal *Laminaria digitata* after experimental spring and autumn removal. *Cahiers De Biologie Marine*, 52 (4), 441-453.

FILBEE-DEXTER, K. AND SMAJDOR, A. 2019. Ethics of assisted evolution in marine conservation. *Frontiers in Marine Science*, 6, 20.

FREDRIKSEN, S., SJØTUN, K., LEIN, T.E. AND RUENESS, J. 1995. Spore dispersal in *Laminaria hyperborea* (Laminariales, Phaeophyceae). *Sarsia*, 80 (1), 47-53.

FREDRIKSEN, S., FILBEE-DEXTER, K., NORDERHAUG, K. M., STEEN, H., BODVIN, T., COLEMAN, M. A., MOY, F. AND WERNBERG, T. 2020. Green gravel: a novel restoration tool to combat kelp forest decline. Scientific Reports, 10, 3983.

GLEASON, M.G., CASELLE, J.E., HEADY, W.N., SACCOMANNO, V.R., ZIMMERMAN, J., MCHUGH, T.A. AND EDDY, N. 2021. A structured approach for kelp restoration and management decisions in California. *The Nature Conservancy*, Arlington, Virginia.

GROENEVELD, R., CHEN, W. AND HYNES, S. 2019. Marine Ecosystem Restoration in Changing European Seas MERCES. D7.4: Restoring marine ecosystems cost-effectively: lessons learned from the MERCES project**.**

HOLBROOK, S.J., CARR, M.H., SCHMITT, R.J. AND COYER, J.A. 1990. Effect of giant kelp on local abundance of reef fishes: the importance of ontogenetic resource requirements. *Bulletin of Marine Science*, 47(1), 104-114.

INCE, R., HYNDES, G.A., LAVERY, P.S. AND VANDERKLIFT, M.A. 2007. Marine macrophytes directly enhance abundances of sandy beach fauna through provision of food and habitat. *Estuarine, Coastal and Shelf Science*, 74(1-2), 77-86.

KAIN, J.M. 1979. A view of the genus *Laminaria*. *Oceanography and Marine Biology: an Annual Review*, 17, 101-161.

KRUMHANSL, K.A. AND SCHEIBLING, R.E. 2012. Production and fate of kelp detritus. *Marine Ecology Progress Series*, 467, 281-302.

LAYTON, C., SHELAMOFF, V., CAMERON, M.J., TATSUMI, M., WRIGHT, J.T. AND JOHNSON, C.R. 2019. Resilience and stability of kelp forests: The importance of patch dynamics and environment-engineer feedbacks. *PLoS ONE,* 14(1), e0210220.

Layton, C., Coleman, M.A., Marzinelli, E.M., Steinberg, P.D., Swearer, S.E., Vergés, A., Wernberg, T. and Johnson, C.R. 2020. Kelp forest restoration in Australia. *Frontiers in Marine Science*, 7, 74.

LEINAAS, H.P. AND CHRISTIE, H. 1996. Effects of removing sea urchins (*Strongylocentrotus droebachiensis*): stability of the barren state and succession of kelp forest recovery in the east Atlantic. *Oecologia*, **105**(4), 524-536.

LÜNING, K. 1990. *Seaweeds: their environment, biogeography, and ecophysiology*. John Wiley AND Sons.

MORRIS, R.L., HALE, R., STRAIN, E.M., REEVES, S.E., VERGÉS, A., MARZINELLI, E.M., LAYTON, C., SHELAMOFF, V., GRAHAM, T.D., CHEVALIER, M. AND SWEARER, S.E. 2020.

Key principles for managing recovery of kelp forests through restoration. *BioScience*, 70(8), 688-698.

NORTH, W.J. 1978. Evaluation, management, and cultivation of Macrocystis kelp forests. California Inst. of Tech., Pasadena, CA (USA). WM Keck Lab. of Engineering Materials.

NORTON, T.A. 1992. Dispersal by macroalgae. *British Phycological Journal*, 27, 293-301.

OYAMADA, K., TSUKIDATE, M., WATANABE, K., TAKAHASHI, T., ISOO, T. AND TERAWAKI, T. 2008. A field test of porous carbonated blocks used as artificial reef in seaweed beds of *Ecklonia cava*. *Journal of Applied Phycology*, 20, 863-868.

PHILIPPART, C.J., ANADÓN, R., DANOVARO, R., DIPPNER, J.W., DRINKWATER, K.F., HAWKINS, S.J., OGUZ, T., O'SULLIVAN, G. AND REID, P.C. 2011. Impacts of climate change on European marine ecosystems: observations, expectations and indicators. *Journal of Experimental Marine Biology and Ecology*, 400 (1), 52-69.

SAUNDERS, M.I., DOROPOULOS, C., BAYRAKTAROV, E., BABCOCK, R.C., GORMAN, D., EGER, A.M., VOZZO, M.L., GILLIES, C.L., VANDERKLIFT, M.A., STEVEN, A.D. AND BUSTAMANTE, R.H. 2020, Bright Spots in Coastal Marine Ecosystem Restoration, *Current Biology*, 30(24), R1500-R1510.

SMALE, D.A., BURROWS, M.T., MOORE, P., O'CONNOR, N. AND HAWKINS, S.J. 2013. Threats and knowledge gaps for ecosystem services provided by kelp forests: a northeast Atlantic perspective. *Ecology and evolution*, 3(11), 4016-4038.

SMALE, D.A., WERNBERG, T., YUNNIE, A.L. AND VANCE, T. 2015. The rise of *Laminaria ochroleuca* in the Western English Channel (UK) and comparisons with its competitor and assemblage dominant Laminaria *hyperborea*. *Marine ecology*, 36(4), 1033-1044.

SMITH, B.D. 1985. Recovery following experimental harvesting of *Laminaria longicruris* and *Laminaria digitata* in Southwestern Nova Scotia. *Helgolander Meeresuntersuchungen*, 39(1), 83-101.

SOUTHERN CALIFORNIA EDISON 2017. *Rebuttal Testimony of Southern California Edison in Support of the Joint Application for Cost Recovery of the Wheeler North Reef Expansion Project Marine Mitigation Costs.* Rosemead, CL: Southern California Edison.

STENECK, R.S., GRAHAM, M.H., BOURQUE, B.J., CORBETT, D., ERLANDSON, J.M., ESTES, J.A. AND TEGNER, M.J. 2002. Kelp forest ecosystems: biodiversity, stability, resilience and future. *Environmental conservation*, 29(4), 436-459.

SUNDENE, O. 1962. The implications of transplant and culture experiments on the growth and distribution of *Alaria esculenta*. *Nytt Magasin for Botanik*, 9, 155-174.

TEAGLE, H., HAWKINS, S.J., MOORE, P.J. AND SMALE, D.A. 2017. The role of kelp species as biogenic habitat formers in coastal marine ecosystems. *Journal of Experimental Marine Biology and Ecology*, 492, 81-98.

TICKELL, S.C.Y., SÁENZ-ARROYO, A. AND MILNER-GULLAND, E.J. 2019. Sunken worlds: The past and future of human-made reefs in marine conservation. *BioScience*, 69(9), 725-735.

TÜRKER, U., YAGCI, O. AND KABDAŞLI, M.S. 2006. Analysis of coastal damage of a beach profile under the protection of emergent vegetation. *Ocean Engineering*, 33(5-6), 810-828.

TURNER, R.E. AND BOYER, M.E. 1997. Mississippi River diversions, coastal wetland restoration/creation and an economy of scale. *Ecological Engineering*, 8(2), 117-128.

VERDURA, J., SALES, M., BALLESTEROS, E., CEFALÌ, M.E. AND CEBRIAN, E. 2018. Restoration of a Canopy-Forming Alga Based on Recruitment Enhancement: Methods and Long-Term Success Assessment. *Frontiers in Plant Science,* 9, 1832.

Appendix 11 Maerl bed and maerl communities (redistribution of dead maerl)

Mitigation to support natural recovery. In order to allow channel dredging in areas with live maerl (Fal and Helford) mitigation options have included the use of a back hoe dredger rather than suction dredging (to reduce sediment suspension and resultant siltation) (Hall-Spencer, undated letter).

Assisted recovery of species assemblage associated with maerl. It was suggested for the Fal and Helford mitigation that coarse maerl gravel (dead maerl) be redistributed (using a back hoe dredger) over a 6 hectare area of seabed in a 1 m thick layer to allow recolonization of maerl communities (which inhabit depths to 70 cm) to assist recovery of maerl and maerl bed communities (Hall-Spencer, undated letter).

An experimental trial to mitigate dredging impact was undertaken within Falmouth Harbour, UK. A surface layer of dead maerl was removed for storage on a barge and the channel was allowed to be deepened before re-laying the maerl. The resilience (resistance and recovery) of the habitat and faunal assemblage to this disturbance was assessed. Maerl (25 m^2 plots, top 0.3 m) was removed, stored on a barge and re-laid by backhoe dredger. Following the mechanical disturbance, the maerl matrix structure was altered through loss of fine sediment from the lower half of cores (>10 cm). There was also a significant reduction in the number of taxa and abundance of infauna and a change in the assemblage composition. By week 44, however, no such significant differences were evident, indicating that the infauna was in a state of recovery. The only response variable showing recovery was annelid biomass. The trial demonstrated that removing and re-laying the top 0.3 m of maerl habitat is technically feasible, and whilst some differences in the habitat structure following re-laying were evident, this did not affect the habitat quality enough to prevent recolonization of infauna. However, this study does not apply to live maerl or long-lived species associated with maerl (Sheehan and others 2015).

Habitats recovery option is applicable to:

EUNIS A5.51 Maerl beds

Natural recovery summary for applicable habitats:

The MarLIN sensitivity assessments for maerl beds contain detailed information on recovery: this section provides a brief summary of the information that the MarLIN assessment (Perry and Tyler-Walters, 2018) provides.

Individual maerl thalli may live for >100 years (Foster, 2001). The maerl bed at St Mawes Bank, Falmouth is estimated to have a maximum age of 4000 years (Bosence and Wilson, 2003) while a maerl bed in the Sound of Iona is up to 4000 years old (Hall-Spencer and others, 2003). Maerl is highly sensitive to damage from any source due to this very slow rate of growth (Hall-Spencer, 1998). Maerl is also very slow to recruit as it rarely produces reproductive spores.

Maerl has a complex three-dimensional structure with interlocking thalli providing a wide range of niches for infaunal and epifaunal invertebrates (Birkett and others, 1998).The interstitial space provided by maerl beds allow water to flow through the bed, and oxygenated water to penetrate at depth so that other species can colonize the bed to greater depths than most other sediments.

Many maerl populations are small either naturally or following disturbance. This reduced brood stock size is an important bottle-neck for species of maerl due to their dependence on fragmentation for propagation. The lack of sexual reproduction further reduces the recruitment potential and dispersal capability. The most important factor in hampering the recovery of maerl is the extremely slow rate of growth. Traditional restoration techniques of brood stock enhancement, hatchery production of adults and habitat creation all appear to be incompatible with the limiting factors described above. Recovery, if possible, is likely to be on the same time-scale as maerl bed turn-over and accumulation, that is, measured in the hundreds or thousands of years.

Maerl bed communities occur on both live and dead maerl and if left undisturbed, may potentially recover regardless of the low reproductive capacity of maerl. This assumes that the integrity of the remaining maerl has been maintained (or can recover) to support an associated community (see feasibility and risks below). Experts predict that the populations of large burrowers associated with maerl (heart urchins, thalassinid shrimps and sessile bivalves such as *Dosinia exoleta* and *Mya truncate)* would take 20-50 years to recover age structure based on longevity (Hall-Spencer, undated letter).

Overview of approach advantages when compared to natural recovery

Natural recovery of maerl beds is prolonged due to low growth rates. No evidence is available to assess whether this approach is successful and supports quicker recovery, but this is likely given the slow rate of establishment of maerl beds.

Overview of approach disadvantages compared to natural recovery

Translocation of maerl would kill the organisms within it and disrupt the structure (Hall-Spencer, undated letter).Recovery options refer to translocation of maerl within adjacent habitats, therefore no impacts on genetic diversity are expected.

Maerl associated communities: Economic costs

No economic costs were identified. The following information provides examples of work required but are not exhaustive and Objective 4 outlines best practice around work plans.

Capital costs: Time should be allocated to site selection to ensure conditions are matched (particularly sediments to prevent smothering and siltation).

Operation costs: Translocation of dead maerl: costs include ship (dredger time) to harvest, store and relocate the maerl. This is a relatively low cost option that does not involve hatcheries etc, infrastructure or maintenance.

Maintenance: Ongoing maintenance not required.

Monitoring of recovery will require vessel and scuba time for sampling (epifauna and infaunal coring) or drop down camera.

Maerl associated communities: feasibility

Translocation of dead maerl gravel requires that the translocation site characteristics match the donor site to prevent sediment changes and to allow characteristic infauna to recolonize. A key factor is to ensure the maerl is not covered or silted otherwise recovery will be prevented. The feasibility of this approach to support the recovery of maerl and associated long-lived species has not been assessed.

Maerl associated communities: risks, challenges and uncertainties

Risks: Storage of dead maerl on barges as proposed in the Fal and Helford may result in anoxia in associated sediments as infauna die and decompose inhibiting recolonization (Hall-Spencer, letter, undated). Inundation of coarse maerl gravel by other sediment reduces suitability for maerl infauna.

Uncertainties: success rates for maerl and long-lived species have not been assessed.

Recommendations for further work

Conservation of maerl beds *in-situ* is preferable to assisted recovery, however, if impacts on maerl beds are unavoidable, projects to assess the effectiveness of this approach coupled with long-term monitoring would allow feasibility and success rates to be better understood.

Maerl assisted recovery: references

BIRKETT, D.A., MAGGS, C.A. AND DRING, M.J., 1998. Maerl. an overview of dynamic and sensitivity characteristics for conservation management of marine SACs. Natura 2000 report prepared by Scottish Association of Marine Science (SAMS) for the UK Marine SACs Project., Scottish Association for Marine Science. (UK Marine SACs Project, vol V.).

BOSENCE D. AND WILSON J. 2003. Maerl growth, carbonate production rates and accumulation rates in the northeast Atlantic. Aquatic Conservation: Marine and Freshwater Ecosystems, 13, S21-S31.

FOSTER, M.S., 2001. Rhodoliths: between rocks and soft places. Journal of Phycology, 37 (5), 659-667.

HALL-SPENCER, J.M., 1998. Conservation issues relating to maerl beds as habitats for molluscs. Journal of Conchology Special Publication, 2, 271-286.

HALL-SPENCER, J.M., GRALL, J., MOORE, P.G. AND ATKINSON, R.J.A., 2003. Bivalve fishing and maerl-bed conservation in France and the UK - retrospect and prospect. Aquatic Conservation: Marine and Freshwater Ecosystems, 13, Suppl. 1 S33-S41.

HALL-SPENCER undated letter. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/332428/maerl_recovery_report.pdf

PERRY, F. AND TYLER-WALTERS, H., 2018. Maerl beds. In Tyler-Walters H. Marine Life Information Network: Biology and Sensitivity Key Information Reviews, [on-line]. Plymouth: Marine

Biological Association of the United Kingdom. [cited 13-03-2022]. Available from: https://www.marlin.ac.uk/habitat/detail/255

SHEEHAN, E.V., BRIDGER, D., COUSENS, S.L. AND ATTRILL, M.J., 2015. Testing the resilience of dead maerl infaunal assemblages to the experimental removal and re-lay of habitat. Marine Ecology Progress Series, 535, pp.117-128.

Bibliography

ABELSON, A., HALPERN, B.S., REED, D.C., ORTH, R.J., KENDRICK, G.A., BECK, M.W., BELMAKER, J., KRAUSE, G., EDGAR, G.J., AIROLDI, L. AND BROKOVICH, E., 2016. Upgrading marine ecosystem restoration using ecological-social concepts. BioScience, 66(2), pp.156-163.

ABPMER, 2020. Mitigation and Compensation Opportunity in Marine Consenting, ABPmer Report No. R.3385. A report produced by ABPmer for Welsh Government, March 2020.

ALMEIDA, M., RAPOSO, A.C., GUERRA, M.T., GAUDÊNCIO, M.J., MARANHÃO, P., RODRIGUES, N.V. AND LEANDRO, S., 2016. Macrobenthic assemblages in an artificial reef in central-west Portugal: Preliminary results.

BAGGETT, L.P., POWERS, S.P., BRUMBAUGH, R.D., COEN, L.D., DEANGELIS, B.M., GREENE, J.K., HANCOCK, B.T., MORLOCK, S.M., ALLEN, B.L., BREITBURG, D.L. AND BUSHEK, D., 2015. Guidelines for evaluating performance of oyster habitat restoration. Restoration Ecology, 23(6), pp.737-745.

BASCONI, L., CADIER, C. AND GUERRERO-LIMÓN, G. 2020. Challenges in marine restoration ecology: how techniques, assessment metrics, and ecosystem valuation can lead to improved restoration success. In YOUMARES 9-The Oceans: Our Research, Our Future (pp. 83-99). Springer, Cham.

BAYRAKTAROV, E., SAUNDERS, M.I., ABDULLAH, S., MILLS, M., BEHER, J., POSSINGHAM, H.P., MUMBY, P.J. AND LOVELOCK, C.E., 2016. The cost and feasibility of marine coastal restoration. Ecological Applications, 26(4), pp.1055-1074.

BERTOCCI, I., ARAÚJO, R., OLIVEIRA, P. AND SOUSA-PINTO, I. 2015. Potential effects of kelp species on local fisheries. Journal of Applied Ecology, 52(5), 1216-1226.

BORJA, A., DAUER, D.M. and GRÉMARE, A. 2012.The importance of setting targets and reference conditions in assessing marine ecosystem quality. Ecological Indicators, 12, 1–7

BORJA, Á., DAUER, D.M., ELLIOTT, M. AND SIMENSTAD, C.A., 2010. Medium-and long-term recovery of estuarine and coastal ecosystems: patterns, rates and restoration effectiveness. Estuaries and Coasts, 33(6), pp.1249-1260.

BROMLEY, C., MCGONIGLE, C., ASHTON, E.C. AND ROBERTS, D., 2016. Restoring degraded European native oyster, *Ostrea edulis*, habitat: is there a case for harrowing?. Hydrobiologia, 768(1), pp.151-165.

CALLIHAN, R., DEPRO, B., LAPIDUS, D., SARTWELL, T. AND VIATOR, C. 2016. Economic analysis of the costs and benefits of restoration and enhancement of shellfish habitat and oyster propagation in North Carolina. RTI International.

CAMPBELL, A.H., MARZINELLI, E.M., VERGES, A., COLEMAN, M.A. AND STEINBERG, P.D. 2014. Towards restoration of missing underwater forests. *PloS one*, *9*(1), p.e84106.

CAPELLE, J.J., WIJSMAN, J.W., SCHELLEKENS, T., VAN STRALEN, M.R., HERMAN, P.M. AND SMAAL, A.C., 2014. Spatial organisation and biomass development after relaying of mussel seed. Journal of Sea Research, 85, pp.395-403.

CASOLI, E., VENTURA, D., MANCINI, G., CARDONE, S., FARINA, F., DONNINI, L., PACE, D.S., SHAUL, R., BELLUSCIO, A. AND ARDIZZONE, G. 2022. Rehabilitation of Mediterranean animal forests using gorgonians from fisheries by-catch. Restoration Ecology, 30(1), p.e13465.

CLAISSE, J.T., WILLIAMS, J.P., FORD, T., PONDELLA, D.J., MEUX, B. AND PROTOPAPADAKIS, L., 2013. Kelp forest habitat restoration has the potential to increase sea urchin gonad biomass. Ecosphere, 4(3), pp.1-19.

CONNELL, J.H. AND SOUSA, W.P., 1983. On the evidence needed to judge ecological stability or persistence. The American Naturalist, 121(6), pp.789-824.

COOPER, K., BURDON, D., ATKINS, J.P., WEISS, L., SOMERFIELD, P., ELLIOTT, M., TURNER, K., WARE, S. AND VIVIAN, C., 2013. Can the benefits of physical seabed restoration justify the costs? An assessment of a disused aggregate extraction site off the Thames Estuary, UK. Marine pollution bulletin, 75(1-2), pp.33-45.

COOPER, K., WARE, S., VANSTAEN, K. AND BARRY, J., 2011. Gravel seeding–A suitable technique for restoring the seabed following marine aggregate dredging?. Estuarine, Coastal and Shelf Science, 91(1), pp.121-132.

CORREIA, M., PALMA, J., KOLDEWEY, H. AND ANDRADE, J.P., 2013. Can artificial holdfast units work as a habitat restoration tool for long-snouted seahorse (Hippocampus guttulatus Cuvier)?. Journal of Experimental Marine Biology and Ecology, 448, pp.258-264.

DAFFORN, K.A., GLASBY, T.M., AIROLDI, L., RIVERO, N.K., MAYER-PINTO, M. AND JOHNSTON, E.L., 2015. Marine urbanization: an ecological framework for designing multifunctional artificial structures. Frontiers in Ecology and the Environment, 13(2), pp.82-90.

DANOVARO, R., ARONSON, J., CIMINO, R., GAMBI, C., SNELGROVE, P.V. AND VAN DOVER, C., 2021. Marine ecosystem restoration in a changing ocean. Restoration Ecology, 29, p.e13432.

DE GROOT, R.S., BLIGNAUT, J., VAN DER PLOEG, S., ARONSON, J., ELMQVIST, T. AND FARLEY, J., 2013. Benefits of investing in ecosystem restoration. Conservation Biology, 27(6), pp.1286-1293.

DE PAOLI, H., VAN DE KOPPEL, J., VAN DER ZEE, E., KANGERI, A., VAN BELZEN, J., HOLTHUIJSEN, S., VAN DEN BERG, A., HERMAN, P., OLFF, H. AND VAN DER HEIDE, T., 2015. Processes limiting mussel bed restoration in the Wadden-Sea. Journal of Sea Research, 103, pp.42-49.

DERNIE, K. M., M. J. KAISER, AND R. M. WARWICK. 2003. Recovery rates of benthic communities following physical disturbance. Journal of Animal Ecology 72 (6), 1043-1056.

DOBSON, A.P., BRADSHAW, A.D. AND BAKER, A.Á.1997. Hopes for the future: restoration ecology and conservation biology. *Science*, *277*(5325), pp.515-522.

DUARTE, C.M., AGUSTI, S., BARBIER, E., BRITTEN, G.L., CASTILLA, J.C., GATTUSO, J.P., FULWEILER, R.W., HUGHES, T.P., KNOWLTON, N., LOVELOCK, C.E. AND LOTZE, H.K., 2020. Rebuilding marine life. Nature, 580(7801), 39-51.

DUARTE, C.M., BORJA, A., CARSTENSEN, J., ELLIOTT, M., KRAUSE-JENSEN, D. AND MARBÀ, N., 2015. Paradigms in the recovery of estuarine and coastal ecosystems. Estuaries and Coasts, 38(4),1202-1212.

EDWARDS, P.E.T., SUTTON-GRIER, A.E. AND COYLE, G.E., 2013. Investing in nature: restoring coastal habitat blue infrastructure and green job creation. Marine Policy, 38, 65-71.

EGER, A., MARZINELLI, E., CHRISTIE, H., FUJITA, D., HONG, S., KIM, J.H., LIEE, L.C., MCHUGH, T.A., NISHIHARA, G.N., VASQUEZ, A.P.G. AND TATSUMI, M. 2021. Global Kelp Forest Restoration: Past lessons, status, and future goals. https://doi.org/10.32942/osf.io/emaz2

ELLIOTT, M. AND CUTTS, N.D., 2004. Marine habitats: loss and gain, mitigation and compensation. Marine Pollution Bulletin, 9(49), pp.671-674.

ELLIOTT, M., BURDON, D., HEMINGWAY, K.L. AND APITZ, S.E., 2007. Estuarine, coastal and marine ecosystem restoration: confusing management and science–a revision of concepts. Estuarine, Coastal and Shelf Science, 74(3), pp.349-366.

ELLIOTT, M., MANDER, L., MAZIK, K., SIMENSTAD, C., VALESINI, F., WHITFIELD, A. AND WOLANSKI, E., 2016. Ecoengineering with ecohydrology: successes and failures in estuarine restoration. Estuarine, Coastal and Shelf Science, 176, pp.12-35.

ELLIS, C.D., HODGSON, D.J., DANIELS, C.L., BOOTHROYD, D.P., BANNISTER, R.C.A. AND GRIFFITHS, A.G., 2015. European lobster stocking requires comprehensive impact assessment to determine fishery benefits. ICES Journal of Marine Science, 72(suppl_1), pp.i35-i48.

FARIÑAS-FRANCO, J.M. AND ROBERTS, D., 2014. Early faunal successional patterns in artificial reefs used for restoration of impacted biogenic habitats. Hydrobiologia, 727(1), pp.75-94.

FIELD, C. D. 1998. Rehabilitation of mangrove ecosystems: an overview. Marine Pollution Bulletin 37:383–392.

FONNER, R., HONEA, J., JORGENSEN, J.C., PLUMMER, M. AND MCCLURE, M., 2021. Considering intervention intensity in habitat restoration planning: An application to Pacific salmon. Journal of Environmental Management, 299, p.113536.

FREDRIKSEN, S., FILBEE-DEXTER, K., NORDERHAUG, K. M., STEEN, H., BODVIN, T., COLEMAN, M. A., MOY, F. AND WERNBERG, T. 2020. Green gravel: a novel restoration tool to combat kelp forest decline. Scientific Reports, 10, 3983.

GAMBLE, C., GLOVER, A., DEBNEY, A., BERTELLI, C., GREEN, B., HENDY, I., LILLEY, R., NUUTTILA, H., POTOUROGLOU, M., RAGAZZOLA, F. AND UNSWORTH, R., 2021, November. Seagrass Restoration Handbook: UK and Ireland. Zoological Society of London.

GEIST, J. AND HAWKINS, S.J., 2016. Habitat recovery and restoration in aquatic ecosystems: current progress and future challenges. Aquatic Conservation: Marine and Freshwater Ecosystems, 26(5), pp.942-962.

GHOSH, A.K., PATTNAIK, A.K. AND BALLATORE, T.J., 2006. Chilika Lagoon: Restoring ecological balance and livelihoods through re-salinization. Lakes & Reservoirs: Research & Management, 11(4), pp.239-255.

GWIN, S.E., M.E. KENTULA, AND P.W. SHAFFER. 1999. Evaluating the Effects of Wetland Regulation through Hydrogeomorphic Classification and Landscape Profiles. Wetlands. Vol. 19 (3). p. 477-489.

HAWKINS, S.J., O'SHAUGHNESSY, K.A., ADAMS, L.A., LANGSTON, W.J., BRAY, S., ALLEN, J.R., WILKINSON, S., BOHN, K., MIESZKOWSKA, N. AND FIRTH, L.B., 2020. Recovery of an urbanised estuary: Clean-up, de-industrialisation and restoration of redundant dock-basins in the Mersey. Marine Pollution Bulletin, 156, p.111150.

HISCOCK, K., BAYLEY, D., PADE, N., LACEY, C., COX, E. AND ENEVER, R., 2013. Prioritizing action for recovery and conservation of marine species: a case study based on species of conservation importance around England. Aquatic Conservation: Marine and Freshwater Ecosystems, 23(1), pp.88-110.

HOLLING, C., 1986. The resilience of terrestrial ecosystems; local surprise and global change. Sustainable Development of the Biosphere, pp.292-317.

HYNES, S., BURGER, R., TUDELLA, J., NORTON, D. AND CHEN, W. 2022. Estimating the costs and benefits of protecting a coastal amenity from climate change-related hazards: Nature based solutions via oyster reef restoration versus grey infrastructure. Ecological Economics, 194, p.107349.

JENSEN, A. C. 2002. Artificial reefs of Europe: perspective and future. – ICES Journal of Marine Science, 59: S3–S13

JONES, B.L., UNSWORTH, R.K., MCKENZIE, L.J., YOSHIDA, R.L. AND CULLEN-UNSWORTH, L.C. 2018. Crowdsourcing conservation: The role of citizen science in securing a future for seagrass. Marine pollution bulletin, 134, 210-215.

KAISER, M.J., CLARKE, K.R., HINZ, H., AUSTEN, M.C., SOMERFIELD, P.J. AND KARAKASSIS, I., 2006. Global analysis of response and recovery of benthic biota to fishing. Marine Ecology Progress Series, 311, pp.1-14.

KENT, F., LILLEY, R., UNSWORTH, R., CUNNINGHAM, S., BEGG, T., BOULCOTT, P., JEORRETT, C., HORSBURGH, R. AND MICHELOTTI, M. 2021. NatureScot Research Report 1286-Seagrass restoration in Scotland-handbook and guidance.

LAYTON, C., COLEMAN, M.A., MARZINELLI, E.M., STEINBERG, P.D., SWEARER, S.E., VERGÉS, A., WERNBERG, T. AND JOHNSON, C.R. 2020. Kelp forest restoration in Australia. Frontiers in Marine Science, 7, 74.

LEFCHECK, J.S., HUGHES, B.B., JOHNSON, A.J., PFIRRMANN, B.W., RASHER, D.B., SMYTH, A.R., WILLIAMS, B.L., BECK, M.W. AND ORTH, R.J. 2019. Are coastal habitats important nurseries? A meta-analysis. Conservation Letters, 12(4), p.e12645.

LINARES, C., GORI, A., CARREIRO-SILVA, M., MONTSENY M, BILAN M, AFONSO, P. FONTES J., GODINHO A., RAKKA, M., MARTINS, I., MORATO, T. D4. 3. 2020 Effectiveness of tools/techniques for restoration in the deep-sea Marine Ecosystem Restoration in Changing European Seas MERCES.

LIVERSAGE, K., 2020. An example of multi-habitat restoration: Conceptual assessment of benefits from merging shellfish-reef and boulder-reef restorations. Ecological Engineering, 143, p.105659.

LOTZE, H.K., COLL, M., MAGERA, A.M., WARD-PAIGE, C. and AIROLDI, L., 2011. Recovery of marine animal populations and ecosystems. Trends in ecology and evolution, 26(11), pp.595-605.

MARINE ECOLOGICAL SURVEYS LIMITED (MES), 2008. Marine Macrofauna Genus Trait Handbook. Marine Ecological Surveys Limited: Bath.

MARION, S.R. AND ORTH, R.J., 2010. Innovative Techniques for large-scale seagrass restoration using Zostera marina (eelgrass) seeds. Restoration Ecology, 18(4), pp.514-526.

MAZIK, K., STRONG, J., LITTLE, S., BHATIA, N., MANDER, L., BARNARD, S. AND ELLIOTT, M. 2015. A review of the recovery potential and influencing factors of relevance to the management of habitats and species within Marine Protected Areas around Scotland. Scottish Natural Heritage Commissioned Report No. 771.Commissioned Report No. 771.

MMO 2019. Identifying sites suitable for marine habitat restoration or creation. A report produced for the Marine Management Organisation by ABPmer and AER, MMO Project No: 1135, February 2019, 93pp

MONTSENY, M., LINARES, C., VILADRICH, N., CAPDEVILA, P., AMBROSO, S., DÍAZ, D., GILI, J.M. AND GORI, A., 2020. A new large-scale and cost-effective restoration method for cold-water coral gardens. Aquatic Conservation: Marine and Freshwater Ecosystems, 30(5), pp.977-987.

NATURESCOT. 2022. Information Note - Marine Habitat Enhancement, Recovery, Restoration and Creation in Scotland: Terminology and Examples. Available on-line: https://www.nature.scot/doc/information-note-marine-habitat-enhancement-recovery-restoration-and-creation-scotland-terminology

NYSTRÖM, M., NORSTRÖM, A.V., BLENCKNER, T., DE LA TORRE-CASTRO, M., EKLÖF, J.S., FOLKE, C., ÖSTERBLOM, H., STENECK, R.S., THYRESSON, M. AND TROELL, M., 2012. Confronting feedbacks of degraded marine ecosystems. Ecosystems, 15(5), pp.695-710.

NYSTRÖM, M., NORSTRÖM, A.V., BLENCKNER, T., DE LA TORRE-CASTRO, M., EKLÖF, J.S., FOLKE, C., ÖSTERBLOM, H., STENECK, R.S., THYRESSON, M. AND TROELL, M., 2012. Confronting feedbacks of degraded marine ecosystems. Ecosystems, 15(5), 695-710.

ONCKEN, N.S., LANGE, T., KRISTENSEN, E., QUINTANA, C.O., STEINFURTH, R.C. AND FLINDT, M.R., 2022. Sand-capping–A large-scale approach to restore organic-enriched estuarine sediments. Marine environmental research, 173, p.105534.

OUNANIAN, K., CARBALLO-CÁRDENAS, E., VAN TATENHOVE, J.P., DELANEY, A., PAPADOPOULOU, K.N. AND SMITH, C.J. 2018. Governing marine ecosystem restoration: the role of discourses and uncertainties. Marine Policy, 96, pp.136-144.

PERROW, M.R. AND DAVY, A.J. eds., 2002. Handbook of ecological restoration (Vol. 2). Cambridge University Press.

PETERSON, G., 2000. Political ecology and ecological resilience: An integration of human and ecological dynamics. Ecological economics, 35(3), pp.323-336.

PIMM, S.L., 1984. The complexity and stability of ecosystems. Nature, 307(5949), pp.321-326.

PONDELLA, D.J., WILLIAMS, J.P., WILLIAMS, C.M., CLAISSE, J.T. AND WITTING, D., 2018. Restoring a Nearshore Rocky Reef Ecosystem in the Challenge of an Urban Setting. In American Fisheries Society Symposium (Vol. 86, pp. 165-186).

POTTS, T., BURDON, D., JACKSON, E., ATKINS, J., SAUNDERS, J., HASTINGS, E. AND LANGMEAD, O. 2014. Do marine protected areas deliver flows of ecosystem services to support human welfare? Marine Policy, 44, pp.139-148.

POWERS SP, BOYER KE 2014. Marine restoration ecology. In: Bertness MD, Bruno JF, Silliman BR, Stachowicz JJ (eds) Marine community ecology and conservation. Sinauer Associates, Sunderland, MA, p 495−516

PRESTON J., GAMBLE, C., DEBNEY, A., HELMER, L., HANCOCK, B. AND ZU ERMGASSEN, P.S.E. (eds) 2020. European Native Oyster Habitat Restoration Handbook. The Zoological Society of London, UK., London, UK

PRONKER, A.E., PEENE, F., DONNER, S., WIJNHOVEN, S., GEIJSEN, P., BOSSIER, P. AND NEVEJAN, N.M., 2015. Hatchery cultivation of the common cockle (Cerastoderma edule L.): from conditioning to grow-out. Aquaculture Research, 46(2), pp.302-312.

SAUNDERS, J; FENN, T AND VERNON, J. 2010. A Framework for Evaluating Restoration Requirements Following Marine Aggregate Extraction. MALSF

SAUNDERS, M.I., DOROPOULOS, C., BAYRAKTAROV, E., BABCOCK, R.C., GORMAN, D., EGER, A.M., VOZZO, M.L., GILLIES, C.L., VANDERKLIFT, M.A., STEVEN, A.D. AND BUSTAMANTE, R.H. 2020. Bright spots in coastal marine ecosystem restoration. Current Biology, 30(24), R1500-R1510.

SCHEFFER, M., CARPENTER, S., FOLEY, J.A., FOLKE, C. AND WALKER, B., 2001. Catastrophic shifts in ecosystems. Nature, 413(6856), 591-596.

SCHOTANUS, J., CAPELLE, J.J., PAREE, E., FIVASH, G.S., VAN DE KOPPEL, J. AND BOUMA, T.J., 2020. Restoring mussel beds in highly dynamic environments by lowering environmental stressors. Restoration Ecology, 28(5), pp.1124-1134.

SCIBERRAS, M., HIDDINK, J.G., JENNINGS, S., SZOSTEK, C.L., HUGHES, K.M., KNEAFSEY, B., CLARKE, L.J., ELLIS, N., RIJNSDORP, A.D., MCCONNAUGHEY, R.A. AND HILBORN, R., 2018. Response of benthic fauna to experimental bottom fishing: A global meta-analysis. Fish and Fisheries, 19(4), pp.698-715.ABPMER, 2020. Mitigation and Compensation Opportunity in Marine Consenting, ABPmer Report No. R.3385. A report produced by ABPmer for Welsh Government, March 2020.

SER. 2004. The SER international primer on ecological restoration. Society for Ecological Restoration.

SHEEHAN, E.V., BRIDGER, D., COUSENS, S.L. AND ATTRILL, M.J., 2015. Testing the resilience of dead maerl infaunal assemblages to the experimental removal and re-lay of habitat. Marine Ecology Progress Series, 535, pp.117-128

SIMPSON, S.L., PRYOR, I.D., MEWBURN, B.R., BATLEY, G.E. AND JOLLEY, D., 2002. Considerations for capping metal-contaminated sediments in dynamic estuarine environments. Environmental science & technology, 36(17), pp.3772-3778.

STEWART-SINCLAIR, P.J., KLEIN, C.J., BATEMAN, I.J. AND LOVELOCK, C.E., 2021. Spatial cost–benefit analysis of blue restoration and factors driving net benefits globally. Conservation Biology, 35(6), pp.1850-1860.

STØTTRUP, J.G., DAHL, K., NIEMANN, S., STENBERG, C., REKER, J., STAMPHØJ, E.M., GÖKE, C. AND SVENDSEN, J.C., 2017. Restoration of a boulder reef in temperate waters: Strategy, methodology and lessons learnt. Ecological Engineering, 102, pp.468-482.

TETT, P., GOWEN, R., PAINTING, S., ELLIOTT, M., FOSTER, R., MILLS, D., BRESNAN, E., CAPUZZO, E., FERNANDES, T., FODEN, J., GEIDER, R., GILPIN, L., HUXHAM, M., MCQUATTERS-GOLLOP, A., MALCOLM, S., SAUX-PICART, S., PLATT, T., RACAULT, M.-F., SATHYENDRANATH, S., MOLEN, J.V.D. and WILKINSON, M. 2013. Framework for understanding marine ecosystem health. Marine Ecology Progress Series, 494, 1–27 + suppl. material.

THELEN, B.A. AND THIET, R.K., 2009. Molluscan community recovery following partial tidal restoration of a New England estuary, USA. Restoration Ecology, 17(5), pp.695-703.

THIET, R.K., KIDD, E., WENNEMER, J.M. AND SMITH, S.M., 2014. Molluscan community recovery in a New England back-barrier salt marsh lagoon 10 years after partial restoration. Restoration ecology, 22(4), pp.447-455.

THOM, R.M. AND WELLMAN, K.F. 1996. Planning aquatic ecosystem restoration monitoring programs. BATTELLE SEATTLE RESEARCH CENTER WA.

UNSWORTH, R.K., BERTELLI, C.M., CULLEN-UNSWORTH, L.C., ESTEBAN, N., JONES, B.L., LILLEY, R., LOWE, C., NUUTTILA, H.K. AND REES, S.C., 2019. Sowing the seeds of seagrass recovery using hessian bags. Frontiers in Ecology and Evolution, p.311.

ZU ERMGASSEN, P.S.E., GAMBLE, C., DEBNEY, A., COLSOUL, B., FABRA, M., SANDERSON, W.G., STRAND, Å. AND PRESTON, J. (eds) 2020. European Guidelines on Biosecurity in Native Oyster Restoration. The Zoological Society of London, UK., London, UK

List of abbreviations

EUNIS: European Nature Information System

GBP: British pound sterling

INNS: Invasive Non-Native Species

MarLIN: Marine Life Information Network

MPA: Marine Protected Area

OWF: Offshore Wind Farm

SEK: Swedish krona

UK MHC: UK Marine Habitat Classification

USD: Unites States dollar

Glossary

Active restoration: Management techniques such as transplanting, planting seeds and seedlings, or the construction of artificial habitats are implemented (Perrow and Davy 2002).

Artificial reef: A submerged structure placed on the substratum (seabed) deliberately, to mimic some characteristics of a natural reef (Jensen, 2002).

Compensation: Project level compensatory measures that are required due to the work having an adverse effect on site integrity to one or more Natura sites/ features (NatureScot, 2022)

Creation: Habitat creation can be used in two different contexts – one using more natural approaches and substrata, for example, creating saltmarsh as part of managed realignment work; and one using artificial substrata or creating habitat where it was not historically present, for example, in offshore marine developments (NatureScot, 2022).

Ecosystem restoration: The process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed" (Society of Ecological Restoration, SER 2004).

Ecoengineering: Manipulating the estuarine or coastal system either to restore it from past degradation or to improve its delivery of nature conservation and natural structure and functioning to increase ecosystem goods, services and societal benefits (Elliott and others, 2016).

Enhancement: The modification of specific structural features to increase one or more functions based on management objectives (Gwin and others 1999).

Intervention intensity: Active interventions "intensity is measured as the amount of restoration (for example, trees planted) per specified location and period. Passive interventions -, intensity may translate, for instance, to the stringency of use restrictions, or other habitat protections, enacted in a given location, during a given period (Fonner and others 2021)

Mitigation: Measures envisaged in order to avoid, reduce and, if possible remedy significant adverse effects (EC Directive 85/337 (EIA Directive)).

Mitigation: The act of making any impact less severe, usually relates to a potential plan or project (Elliott and Cutts, 2004).

Natural recovery: The process by which an ecosystem returns to a prior state following the cessation of some impact or alteration (Abelson and others, 2016).

Passive restoration: Focused on removing the impact of environmental stressors such as pollution or poor water quality, which prevent natural recovery of the ecosystems occurring (Perrow and Davy 2002).

Recoverability: The ability of a habitat, community or individual (or individual colony) of species to redress damage sustained as a result of an external factor. (MarLIN glossary).

Recovered/Restored: Of a habitat, is achieved when 'it contains sufficient biotic and abiotic resources to continue its development without further assistance or subsidy (SER, 2004).

Recovery: Is the process of moving to a state that is considered to be ‘recovered (Mazik and others 2015).

Rehabilitation: The replacement of structural or functional characteristics of an ecosystem that have been diminished or lost (Field 1998).

Resilience: The ability of an ecosystem to return to its original state after being disturbed (from Makins, 1991) (cf. 'constancy', 'persistence', 'stability') (MarLIN glossary).

Resistance: The degree to which a variable is changed following perturbation (Pimm, 1984). The tendency to withstand being perturbed from the equilibrium (Connell and Sousa, 1983) (MarLIN glossary).

Restoration: Relates to the active re-creation of conditions, and can interchangeably be used with ‘rehabilitation’ or ‘remediation’, which are sometimes used to indicate less comprehensive or complete restoration actions or those that create novel habitats without natural analogues. Even precisely distinguishing between ‘conservation’ and ‘restoration’ is neither easy nor practical since conservation typically also comprises some kind of action (for example, in terms of captive breeding or active improvements of the habitat (Geist and Hawkins, 2016).

Restoration: The process of establishing or re-establishing a habitat that in time can come to closely resemble a natural condition in terms of structure and function (Baggett and others 2015).

Restoration ecology: The science underlying the concepts and tools needed to restore ecosystems (SER, 2004).

Rewilding: Refers to restoring processes and functions in very large, landscape/ ecosystem scale projects. Scientifically and scale wise, marine habitat enhancement is at too small a scale currently for this term to be applied (Naturescot 2022).

Transplantation: The movement of the species from a donor site where it is still present to another, where there is the need to restore the vanished habitat.

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