

# 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**  
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# COMMISSIONED REPORT

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**Commissioned Report No. 771**

**A review of the recovery potential and  
influencing factors of relevance to the  
management of habitats and species within  
Marine Protected Areas around Scotland**

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## COMMISSIONED REPORT

# Summary

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A review of the recovery potential and influencing factors of relevance to the management of habitats and species within Marine Protected Areas around Scotland

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**Project No: 14654**

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### **Keywords**

Recovery process; environmental and biological processes influencing recovery; Scottish Marine Protected Areas, Management options for recovery.

### **Background**

In order to better understand the recovery process of temperate marine habitats and species, Scottish Natural Heritage commissioned a literature review, leading to a framework for assessing recovery potential, with a particular (but not exclusive) focus on features within Marine Protected Areas (MPAs) which have been identified as having a conservation objective to 'recover' or 'conserve (uncertain)'. This will enable a consistent approach to recovery (including the way it is defined and the goals for recovery) and will help to inform and standardise the SNH approach to managing the species and habitats within MPAs in order to maximise recovery potential.

A number of species and habitats of national and international importance have been identified in the seas around Scotland through the Scottish Marine Protected Areas Project. A number of these are classed as threatened and/or declining under OSPAR, and draft conservation objectives for these features have been presented to the Scottish Government by SNH and JNCC. However, where a 'recovery' conservation objective has been assigned, further development of the potential approaches that could be followed for achieving this objective is required. Approaches under consideration can be characterised as including the reduction or removal of pressures and consideration of the extent over which management might be applied in relation to the extent of the features within MPAs. However, a better understanding of the effectiveness of these, and other, management options will enable SNH to optimise their approach.

### **Main findings**

- A review of the literature indicated a high degree of variation in the use and interpretation of the term 'recovery'. It is proposed that **'recovery' should refer to a process or trajectory and 'recovered' should refer to an end point**. Separate definitions of recovery and recovered have been proposed for individual species, communities (including biogenic species) and habitats.

- Of the species, communities and habitats reviewed, many show limited recovery potential and of those which could recover, the timescale for recovery is likely to be long (tens to hundreds of years). There are examples of recovery, or indications of recovery, having been achieved as a result of pressure removal.
- Factors limiting recovery included anthropogenic influence, together with aspects of the biology of individual species and communities and environmental conditions. Of key importance is the scale of the disturbance, the degree of habitat homogenisation/fragmentation, species removal and the longevity of physical modifications to the habitat. Recovery is reliant on environmental and biological connectivity between populations or species patches and on the suitability of the habitat in terms of substratum type, depth, water quality and sediment quality. For some species (particularly those inhabiting soft sediments), subtle change in the sediment structure, consolidation, stability and chemistry are of importance. Superimposed upon this are the life history traits of the individual species, which may or may not enhance recovery potential. These include reproductive strategy and frequency, growth rate, longevity and dispersal ability, coupled with biological interactions between species which may influence the direction and timescale of the recovery trajectory.
- A brief review of restoration techniques indicates potential for application to a limited number of species, particularly bivalves. However, the long-term success is unknown and restoration should not be attempted at the expense of other management to enable recovery and/or prevent further decline.
- Management should take account of both environmental and biological connectivity and should aim to protect the interaction between populations and sub-populations (e.g. sources and sinks of propagules). Therefore, management should be targeted at both physical and biological processes and management may have to be applied between MPAs and/or beyond MPA boundaries if recovery is to be achieved.
- Management needs to account for direct (near-field) and indirect (far-field) pressures.

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<b>Table of Contents</b>	<b>Page</b>
<b>1. INTRODUCTION</b>	<b>1</b>
1.1 Aims and objectives	2
1.2 Approach	4
1.2.1 Objective 1: Concept of recovery	4
1.2.2 Objective 2: Literature review of evidence for recovery	4
1.2.3 Objective 3: Develop a framework for assessing recovery potential that SNH can use in considering the recovery of marine habitats and species protected within MPAs	5
1.2.4 Objective 4: Make recommendations on how this framework could be applied, where knowledge gaps exist and what further work would improve the understanding of recovery processes	6
1.2.5 Objective 5: Workshop to review and refine the proposed framework for assessing recovery potential	6
<b>2. CONCEPT OF RECOVERY</b>	<b>7</b>
2.1 Introduction	7
2.2 Variability in the definition of recovery	9
2.2.1 Establishing a recovery end point	10
2.2.2 Parameters used to assess recovery and recovered status and study length	11
2.2.3 Determining convergence between recovering and target communities	12
2.2.4 Lifespan or turnover of the feature of interest	13
2.3 Practical definitions of species and habitat 'recovery' and 'recovered' and issues when assessing recovery	13
2.3.1 Over-arching definitions	14
2.3.2 Working definitions	15
2.3.3 Monitoring requirements	16
<b>3. CURRENT KNOWLEDGE OF RECOVERY IN TEMPERATE MARINE ECOSYSTEMS</b>	<b>17</b>
3.1 <i>Limaria hians</i> (flame shell)	17
3.1.1 Threats and sensitivity	17
3.1.2 Overview of relevant studies	18
3.1.3 Factors influencing recovery	19
3.2 <i>Modiolus modiolus</i> (horse mussel)	21
3.2.1 Threats and sensitivity	21
3.2.2 Overview of relevant studies	22
3.2.3 Factors influencing recovery	22
3.3 Maerl beds	27
3.3.1 Threats and sensitivity	27
3.3.2 Overview of relevant studies	27
3.3.3 Factors influencing recovery	28
3.4 Maerl or coarse shell with burrowing sea cucumbers ( <i>Neopentadactyla mixta</i> )	30
3.4.1 Threats and sensitivity	31
3.4.2 Overview of relevant studies	31
3.4.3 Factors influencing recovery	31
3.5 Ocean quahogs ( <i>Arctica islandica</i> )	32
3.5.1 Threats and sensitivity	32
3.5.2 Overview of relevant studies	33
3.5.3 Factors influencing recovery	33
3.6 Soft sediment habitats and communities	34
3.6.1 Burrowed mud	34

3.6.2	Shallow tide-swept coarse sands with burrowing bivalves	35
3.6.3	Overview of relevant studies	36
3.6.4	Factors influencing recovery	37
3.7	Fan mussel aggregations ( <i>Atrina fragilis</i> )	41
3.7.1	Threats and sensitivity	42
3.7.2	Overview of relevant studies	43
3.7.3	Factors influencing recovery	43
3.8	Native oyster ( <i>Ostrea edulis</i> )	44
3.8.1	Threats and sensitivity	45
3.8.2	Overview of relevant studies	45
3.8.3	Factors influencing recovery	46
3.9	Seagrasses ( <i>Zostera</i> species)	49
3.9.1	Threats and sensitivity	49
3.9.2	Overview of relevant studies	50
3.9.3	Factors influencing recovery	50
3.10	Sandeels ( <i>Ammodytes marinus</i> and <i>Ammodytes tobianus</i> )	51
3.10.1	Threats and sensitivity	51
3.10.2	Overview of relevant studies	52
3.10.3	Factors influencing recovery	52
3.11	Common skate ( <i>Dipturus</i> cf. <i>flossada</i> and <i>Dipturus</i> cf. <i>intermedia</i> )	53
3.11.1	Threats and sensitivity	53
3.11.2	Overview of relevant studies	54
3.11.3	Factors influencing recovery	54
3.12	Herring spawning ground	55
3.12.1	Threats and sensitivity	55
3.12.2	Overview of relevant studies	56
3.12.3	Factors influencing recovery	56
3.13	Black guillemot ( <i>Cephus grylle</i> )	57
3.13.1	Threats and sensitivity	58
3.13.2	Overview of relevant studies	59
3.13.3	Factors influencing recovery	60
<b>4.</b>	<b>DEVELOPMENT OF A FRAMEWORK FOR ASSESSING RECOVERY POTENTIAL</b>	<b>62</b>
4.1	Overarching questions relating to the recovery process	62
4.2	Considerations for assessing recovery potential	63
<b>5.</b>	<b>SUMMARY OF FACTORS INFLUENCING RECOVERY</b>	<b>73</b>
5.1	Ecological factors	73
5.1.1	Scale of disturbance and degree of impact	73
5.1.2	Physical recovery of the habitat	74
5.1.3	Habitat fragmentation, homogenisation and connectivity in relation to the hydrodynamic regime	75
5.1.4	Spatial extent, distribution and condition of species and habitats	76
5.1.5	Life history, dispersal ability and recruitment	76
5.1.6	Biological interactions	79
5.2	Management considerations	81
5.2.1	Activities and pressures	81
5.2.2	Size of the management area for MPA features	82
5.3	Active intervention and restoration to improve recovery potential	91
5.3.1	Habitat enhancement	91
5.3.2	Translocation	92
5.3.3	Hatchery cultivation, sea ranching and stocking enhancement	93
<b>6.</b>	<b>KNOWLEDGE GAPS</b>	<b>95</b>
<b>7.</b>	<b>CONCLUSIONS AND RECOMMENDATIONS</b>	<b>97</b>

7.1	Recommendations	100
<b>8.</b>	<b>REFERENCES</b>	<b>103</b>
	<b>ANNEX 1: CONSERVATION OBJECTIVES</b>	<b>123</b>
	<b>ANNEX 2: SEXUAL MATURITY AND LONGEVITY OF SPECIES OF INTEREST IN SCOTTISH WATERS</b>	<b>124</b>
	<b>ANNEX 3: WORKSHOP TO REVIEW AND REFINE THE FRAMEWORK FOR ASSESSING RECOVERY POTENTIAL</b>	<b>126</b>

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## 1. INTRODUCTION

Scottish waters contain a high diversity of species and habitats, some of which have been identified as nationally and internationally important, through the Scottish Marine Protected Areas Project (Baxter *et al.*, 2011; Scottish Natural Heritage, 2012). Under both National and European legislation, the Scottish Government (Marine Scotland) has the ability to establish Marine Protected Areas (MPAs) for the protection of nationally and internationally important features. Relevant legislative drivers for the introduction of MPAs include OSPAR, the EU Habitats and Birds Directives and the Marine Strategy Framework Directive, the Marine and Coastal Access Act 2009, the Marine (Scotland) Act 2010, the Scottish Biodiversity Strategy (SBS) and the Marine Biodiversity Implementation Plan (MBIP) (Howson *et al.*, 2012). The Scottish MPA Project, is one route through which Marine Scotland aims to deliver a 'clean, healthy, safe, productive and biologically diverse marine and coastal environment that meets the long term needs of people and nature' (Scottish Natural Heritage, 2012). Whilst there is a focus on marine conservation, maintaining ecosystem services and societal benefits is also a priority and this may result in a degree of pressure and disturbance to marine species and habitats, although the aim of management is to ensure that this is minimal (Elliott, 2011).

Baxter *et al.* (2011) summarised current trends in the pressures (the mechanism through which an activity affects a biological component of the ecosystem) acting in Scottish waters, together with trends in the status of marine species and habitats. This study indicates that the chemical quality of Scottish waters is largely stable (no apparent trend of increase or decrease), with little concern in most areas. Whilst there are localised concerns relating to marine litter and hazardous substances (generally in the Firths of Forth and Clyde), chemical pressures are generally not considered to be a major, widespread threat to biodiversity. In contrast, Baxter *et al.* (2011) indicates a deterioration in the quality of a number of benthic habitats (and their associated communities) with a high level of concern associated with shallow and subtidal shelf sediments in most areas around Scotland. This deterioration is predominantly linked to physical damage and disturbance. Chaniotis *et al.* (2011) and the Scottish Government (2013) indicate that pressures associated with physical disturbance (ranging from habitat removal to abrasion and alteration of habitat type) are the most prevalent around the Scottish coast. These pressures are largely associated with fishing activities and the use of mobile demersal fishing gear, boat moorings/anchorage, renewable energy development and, in some areas, military activities. Whilst a degree of natural disturbance (physical, chemical and biological) plays a key role in maintaining diversity and community structure (Grassle and Sanders, 1973), high levels of disturbance (natural or anthropogenic) can lead to species loss and complete habitat modification, which may prevent re-colonisation of the lost species. For many species (including species of plankton, mammals, seabirds, water birds, fish, sharks/rays, reptiles/occasional visitors and non-natives), a moderate to high level of concern is associated with most groups.

Disturbance is defined, in ecological terms, as 'an event initiating species population change due to mortality or removal and/or change in the resource base of the community' (Zajac and Whitlatch, 1982). Sousa (1985, in Barnes and Conlan, 2007) later described disturbance to biological systems as 'a discrete, punctuated killing, displacement, or damaging of one or more individuals (or colonies) that directly or indirectly creates an opportunity for new individuals (or colonies) to become established'. Any disturbance acting on a biological element, whether this is at the level of the cell or ecosystem, will firstly be accommodated by 'resistance' to change but then the system will be unable to resist change and will decline (Elliott *et al.*, 2007). The ability of the biological element to recover from that stressor is termed 'resilience' and this is related to the nature of the element and stressor, for example, a habitat adapted to high energy disturbance (e.g. storms) such as a gravel bed will also be more likely to cope with abrasion resulting from anthropogenic activities (e.g. fishing).

In sedimentary environments, disturbance associated with anthropogenic activities can result in sediment removal, smothering, compaction, liquefaction, changes in sediment transport dynamics, changes in particle size distribution/sediment composition and changes in organic content and chemical gradients. In rocky habitats, physical disturbance is more likely to take the form of smothering or abrasion resulting in the direct removal of organisms. Disturbance can result in direct habitat change or changes through shifts in the distribution of key bioengineer species which maintain habitat structure (tubicolous polychaetes, burrowing crustaceans, biogenic reef forming species) (Thrush *et al.*, 2008; Gray and Elliott, 2009). If ineffectively managed, individual and cumulative (i.e. in numerous areas and/or of numerous types) pressures can collectively result in habitat modification or loss, loss of species and loss of functionally important groups (Thrush *et al.*, 2008). This leads to fragmentation (which refers to the size and spatial arrangement of habitats (Ewers, *et al.*, 2006)) and the isolation of habitat patches in which ecological processes differ to those in larger surrounding, continuous habitats (Hovel and Lipcius, 2001). Associated with this isolation is a loss of connectivity and potential for interaction between populations within the isolated habitats (Thrush *et al.*, 2008).

As a result of anthropogenic pressure, possibly combined with climate change and natural population variability, a number of species and habitats (of national and international importance) have been identified as threatened and/or declining, both through local assessment within Scotland and under OSPAR definitions. Draft conservation objectives for the proposed protected features were provided to the Scottish Government by SNH and JNCC ahead of the consultation on possible MPAs (SNH and JNCC, 2013) (Table 1). A conservation objective of 'recover' or 'conserve (uncertain)' has been assigned to a number of these habitats and species and further development of the approach to achieving these objectives is required. Further explanation of these terms and how the conservation objectives were set is outlined in [Annex 1](#). Management options currently under consideration include reduction or removal of pressures and consideration of setting the boundaries of management to support future recovery. Whilst a better understanding of the effectiveness of these, and other, management will enable SNH to optimise their approach, a better understanding of how to determine when recovery has been achieved is required, together with a better understanding of the physical and biological processes which may facilitate or inhibit recovery.

## **1.1 Aims and objectives**

This study aims to examine the evidence for recovery for each of the protected features of MPAs provided by SNH (Table 1), including (where identified) the point at which recovery was declared in the studies examined, the indicators used to define recovery and the timescale for recovery. This will lead to a framework for assessing recovery potential to help inform the approach to achieving recovery in the marine environment ([see section 4.2](#)).

Table 1. Protected features and their prioritisation in this report with regard to their conservation objectives.

<b>Feature</b>
<b>Primary priority for commissioned report –protected features assigned conserve (uncertain) or recover conservation objective</b>
Burrowed mud (including <i>Funiculina quadrangularis</i> , <i>Maxmuelleria lankesteri</i> , <i>Pachycerianthus multiplicatus</i> )
Flame shell beds ( <i>Limaria hians</i> )
Horse mussel beds ( <i>Modiolus modiolus</i> )
Maerl beds
Maerl or coarse shell gravel with burrowing sea cucumbers ( <i>Neopentadactyla mixta</i> )
#Herring spawning grounds ( <i>Clupea harengus</i> )
Shallow tide-swept coarse sands with burrowing bivalves (including <i>Moerella</i> spp.)
Black guillemot ( <i>Cephus grille</i> )
Common skate (Formerly <i>Dipturus batis</i> now split provisionally into <i>D. cf. flossada</i> and <i>D. cf. intermedia</i> )
Sandeels ( <i>Ammodytes marinus</i> , <i>A. tobianus</i> )
Ocean quahog ( <i>Arctica islandica</i> )
<b>Medium priority –protected features of particular interest due to sensitivity but that are assigned a conserve conservation objective</b>
Fan mussel aggregations ( <i>Atrina fragilis</i> )
Native oysters ( <i>Ostrea edulis</i> )
Seagrass beds ( <i>Zostera noltii</i> , <i>Zostera marina/angustifolia</i> , <i>*Ruppia maritima</i> )
*Northern feather star aggregations on mixed substrata ( <i>Leptometra celtica</i> )
*Northern sea fan and sponge communities ( <i>Caryophyllia smithii</i> , <i>Swiftia pallida</i> )
<b>Low priority – features of lower interest due to having a conserve conservation objective, lower sensitivity and/or no possible MPAs being proposed</b>
*Kelp and seaweed communities on sublittoral sediment
*Tide-swept algal communities
*Sea loch egg wrack beds ( <i>Ascophyllum nodosum</i> ecad <i>mackaii</i> )

\* Protected features excluded from the review following a process of prioritisation.

# Herring spawning grounds were a proposed protected feature when this contract commenced, however this feature is not a protected feature as it was not designated as a feature of the Clyde Sea Sill site following decisions made by Marine Scotland after the consultation.

The objectives are:

1. **outline the concept of recovery** and explain its meaning in the context of the MPA project and proposed protected features;
2. **review the literature of the current knowledge of recovery** in the context of temperate marine ecosystems, focusing on protected features for which recover or conserve (uncertain) conservation objectives have been assigned (Table 1);
3. **develop a framework for assessing recovery potential that SNH can use in considering the recovery of marine habitats and species** protected within MPAs. This framework may be used to inform the delineation of an area over which management should be applied in order to facilitate recovery and may include a set of

general considerations together with species and habitat-specific (or groups of species/habitats) considerations;

4. **make recommendations on how this framework could be applied**, where knowledge gaps exist and what further work would improve the understanding of recovery processes. This will provide a framework for guiding future consideration of recovery within Nature Conservation MPAs;
5. **facilitate a workshop** to review and refine the definition of and the framework for assessing recovery potential, in the context of features within MPAs in Scotland.

## 1.2 Approach

### 1.2.1 Objective 1: Concept of recovery

In order for SNH to develop and implement recovery plans, a clear understanding of how to determine whether or not recovery has occurred is essential. As a first step in this project, the work of Mazik and Smyth (2013) was expanded to indicate the key elements of recovery of relevance to most of the habitats and species listed in (Table 1). This stage assessed the parameters and indicators used to assess recovery and the documented timescales over which it has occurred. This enabled:

- identification of the most frequently used set of indicators;
- identification of other information that can be derived from those indicators; and
- proposal of a realistic ‘working’ definition of recovery in the context of the SNH monitoring and assessment programme. This definition was based, in part, on the best available information and will allow for consistent, economically viable and fit-for-purpose monitoring over a realistic timescale.

### 1.2.2 Objective 2: Literature review of evidence for recovery

The review focused on all of the high priority and half of the medium priority species and habitats listed in Table 1 although, where information was lacking (as was the case for many species), comparison was drawn with other species or habitats with similar attributes and environmental requirements. Following a process of prioritisation, the seagrass *Ruppia maritima*, northern feather star aggregations on mixed substrata (*Leptometra celtica*), northern sea fan and sponge communities (*Caryophyllia smithii*, *Swiftia pallida*), kelp and seaweed communities on sublittoral sediment, tide-swept algal communities and *Ascophyllum nodosum* ecad *mackaii* beds were considered to be beyond the scope of this review and were therefore excluded. The primary focus was on impacts and recovery following pressures relevant to Scottish waters, although information on the wider biology of those species listed in Table 1 and of the species found in those habitats listed in Table 1, was also considered. The identification of relevant pressures was based on sensitivity assessments given by the Feature Activity Sensitivity Tool (FEAST) (Scottish Government, 2013). The review phase included the following components and considered the principles of systematic review (CEBC, 2009):

- details of the search strategy, the potential sources of information and synthesis of the evidence;
- a comprehensive literature and data search including (as a minimum) peer reviewed literature, grey literature, internet resources and websites such as MarLIN which gives information on species sensitivity;
- selection and extraction of evidence from relevant articles;
- synthesis and identification of knowledge gaps.

The primary emphasis of the review was on direct evidence for recovery and therefore concentrated on articles containing primary details of experimental work and data with an experimental design to enable objective hypothesis testing and objective analysis. Reviews and conceptual studies (providing indirect evidence and possibly evidence that has not been tested or proven) were used only where objective data were lacking. These studies were clearly identified and any conclusions or recommendations based upon them were treated with caution. Priority was given to peer-reviewed scientific literature and Government Agency reports from both the UK and elsewhere (written by qualified, practising scientists) although it is acknowledged that circumstantial evidence can be valuable where more detailed information is completely lacking. Again, these studies were clearly identified. Biased and emotive articles were not considered.

**Sources of information:**

1. On site collection;
2. Electronic access to a wide range of online articles;
3. Access to the British Library;
4. Access to online literature search facilities (Scopus, Web of Knowledge, Science Direct, SciVerse, Hydra, PubMed, JSTOR, ETHoS)

Other sources of information included Google Scholar and internet searches, Conservation Agency, Environment Agency/Scottish Environment Protection Agency, Marine Scotland/Defra/Cefas, MarLIN online catalogues and discussions with contacts within these organisations. A significant amount of literature is also held in IECS (EndNote Libraries).

*1.2.3 Objective 3: Develop a framework for assessing recovery potential that SNH can use in considering the recovery of marine habitats and species protected within MPAs*

The literature review (Objective 2) highlighted a number of environmental and biological factors that influence the recovery process. These findings were incorporated into a general discussion, drawing on similar evidence from other species and habitats, according to broad categories. These factors were used as a basis to develop a framework for assessing recovery potential. The review identified the quantity and quality of information on recovery processes in temperate marine habitats and identified data/knowledge gaps (Objective 4).

The framework included:

- a set of high level general considerations which could be used for examining recovery potential across the range of protected features for MPAs (Table 1);
- identification of factors that may facilitate or inhibit recovery for specific features.

For features whose conservation objective is recover or conserve (uncertain), SNH require guidance to facilitate their decisions on:

- The spatial area that management should be applied to assist recovery – the protected feature or a wider area.
- Should management for recovery be considered at the scale of the MPA network and wider Scottish Seas in addition to, or as an alternative to, individual MPA management?

SNH anticipate that the framework and the considerations it contains would need to account for:

- Limitations on recovery of the spatial extent of a feature due to complete habitat change or inappropriate neighbouring habitat (e.g. rock biotopes cannot recover or extend into sediment habitats, mixed/coarse sediment communities may not recover in habitats which have become muddy).

- What conditions are suitable for recovery of specific features (Table 1) and for protected features of MPAs in general?
- How close should suitable habitat be to the feature under consideration? For example, can only neighbouring habitats be considered or can suitable habitat further afield be considered? This will relate to dispersal, recruitment and colonisation mechanisms and species mobility.
- Where suitable habitats are in appropriate proximity, how much of the habitat should be incorporated into the MPA boundary? This may be based on previous recovery or restoration studies and growth/recovery rate of the species but also on the availability/status of other suitable habitat (at a regional sea scale) and the degree of threat to a species. For example, if a species was declining throughout Scottish seas but a large area of suitable habitat was available in a specific location, then wide scale protection of that habitat may compensate for lack of suitable habitat elsewhere.
- What spatial scale is appropriate when considering suitable conditions for recovery? For example, over what spatial scale should habitat and water quality be considered and what are the implications for determining suitable management areas and boundaries over which pressure removal or alleviation should be applied?
- Is there any information on connectivity (or habitat fragmentation) that should be taken into account when considering the recovery potential of these features (Table 1)? This includes proximity of a habitat to other patches of the same habitat, species mobility, dispersal mechanisms and ability, recruitment and colonisation mechanisms, longevity of the species, frequency of reproduction and potential for genetic defects associated with inbreeding if habitat patches or populations are insufficiently connected or are of insufficient size.

*1.2.4 Objective 4: Make recommendations on how this framework could be applied, where knowledge gaps exist and what further work would improve the understanding of recovery processes*

Based on the literature review, species and habitat specific knowledge gaps were identified and an assessment of the implications of these gaps for future management was made. This enabled identification of research priorities and the formation of objective recommendations on future work which would improve the ability to manage MPAs and facilitate the recovery of threatened and/or declining species. Recommendations for the application of the framework were made on a feature specific (where possible) and at a higher (more general) level, relating to groups of habitats or species with similar environmental requirements, sensitivities and recovery potential. Additionally, where the information was available, the nature of the stressor, its longevity and the type of footprint it leaves was considered. This included an assessment of practicality and of the spatial and temporal scale at which the framework can be applied.

*1.2.5 Objective 5: Workshop to review and refine the proposed framework for assessing recovery potential*

The framework was refined through its application to a number of case studies (Upper Loch Fyne and Loch Goil, incorporating burrowed mud, maerl, *Arctica islandica*, *Limaria hians* and *Modiolus modiolus*) provided by SNH. This involved a series of presentations (based on the findings of the present report) and group discussions ([Annex 3](#)).

## 2. CONCEPT OF RECOVERY

### 2.1 Introduction

There is a substantial amount of literature addressing recovery in marine and estuarine environments although there is no standardised definition of recovery (Lotze *et al.*, 2011). Elliott *et al.* (2007) implies recovery to be the return of a system to its previous condition following disturbance or degradation. However, in the context of the literature reviewed here, the term recovery has been used interchangeably both as an end point (i.e. having achieved recovery) and as a trajectory or process. Biological assemblages that 'recover' undergo a period of 'recovery' and are best described as 'recovering' (i.e., in the process of moving towards a recovery end point). This separates the end point of a recovered community, habitat or population from the recovery processes that contribute to that end point but do not constitute it. Essentially, **recovery** can be described as the process of returning to a normal state or **recovery endpoint** after some period of being degraded (Borja *et al.*, 2012; Tett *et al.*, 2013) and can be considered as a successional process, facilitated by recruitment and colonisation, which may be described as a set of rate processes (Thrush *et al.*, 2008). **Recovered** status is generally considered to have been achieved when a set of defined recovery endpoints have been achieved. 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. The long-term status of a biological assemblage should be repeatedly monitored to separate the two differing aspects of 'recovery' (trajectory towards a 'recovered' state) and 'recovered' (long term stability at a recognised reference point).

In ecological terms, these concepts can be considered to operate at any level of biological organisation – in the sequence from the cell, tissue, individual, population, community and ecosystem (McLusky and Elliott 2004). It is assumed that degradation, if unchecked, occurs through this sequence and each component needs to recover from a stressor before an ecosystem or any of its major elements (habitat/biotope/community) can be considered to have recovered. Hence, the importance of the precautionary principle which is based on the assumption that a change due to a stressor in a lower level will transfer to upper levels unless prevented. However, it is also acknowledged that the variability within a system confers the ability to absorb stress such that stress at lower levels does not necessarily transfer to higher levels, what may be termed environmental homeostasis (Elliott and Quintino 2007). Some ecosystems with a high natural variability such as estuaries may have a greater ability to absorb such stress (Elliott and Whitfield, 2011).

The recovery process or trajectory is partially dependent on the resistance (ability of a system to absorb pressure without alteration) of the ecological components to change and the resilience of the system, which is defined as 'the ability to return to a state similar to that before the pressure' (Figure 1) (Holling, 1973; Elliott *et al.* 2007). Figure 1 indicates that there may be a time-lag for recovery to occur, following pressure removal, that recovery may follow a different trajectory to that of the change in condition and that a system may return to its original state (complete resilience) or to an alternative state (partial resilience). This model was quantified for an inshore/estuarine system by Borja *et al.* (2010).

Recovery, once disturbance has ceased, is initially dependent on the spatial scale, type, intensity, frequency, duration, and the degree of damage to the habitat (Figure 1; Bell and Devlin, 1983; Thrush *et al.*, 1996; Dernie *et al.*, 2003; Mazik and Smyth, 2013). In order for a community to achieve recovered status, the physical properties of the habitat must reach an appropriate recovery endpoint in terms of (for example) the topography, elevation/depth, particle size and organic content characteristics, consolidation and stability (Dernie *et al.*, 2003). This may also mean improvements in water quality (for both benthic and pelagic species) and recovery (as an endpoint) of components of the benthic system upon which

pelagic species (or other benthic and epibenthic species) may be reliant. Subsequent recruitment and immigration of organisms depends on proximity of a community with a similar structure and connectivity to that community (Santos and Simon, 1980); species mobility, dispersal and supply of propagules (Beukema *et al.*, 1999; Herkul *et al.*, 2011); inter and intra-specific interactions and modification of the sediment through biological activity (Snelgrove and Butman, 1994; Zajac, 2004; Thrush *et al.*, 2008; Volkenborn, 2009). The timing of pressure removal or the implementation of management measures to enable the recovery process to begin can, in some systems, both strongly influence the rate at which recovery can be achieved. For example, recovery that is given the opportunity to start immediately before key spawning periods is likely to be achieved more quickly than if it were not allowed to start until after spawning activity (Beukema *et al.*, 1999). For example, the cessation of dredging in the spring before recruitment occurs may allow more rapid recovery than if it was delayed until after recruitment (Elliott *et al.*, 2007, Cooper *et al.*, 2008).

This section summarises ecological definitions of recovery that have been previously used, considers their objectivity and practicality and suggests a generic, practical working definition of recovery that fits within the scope of the work of SNH.

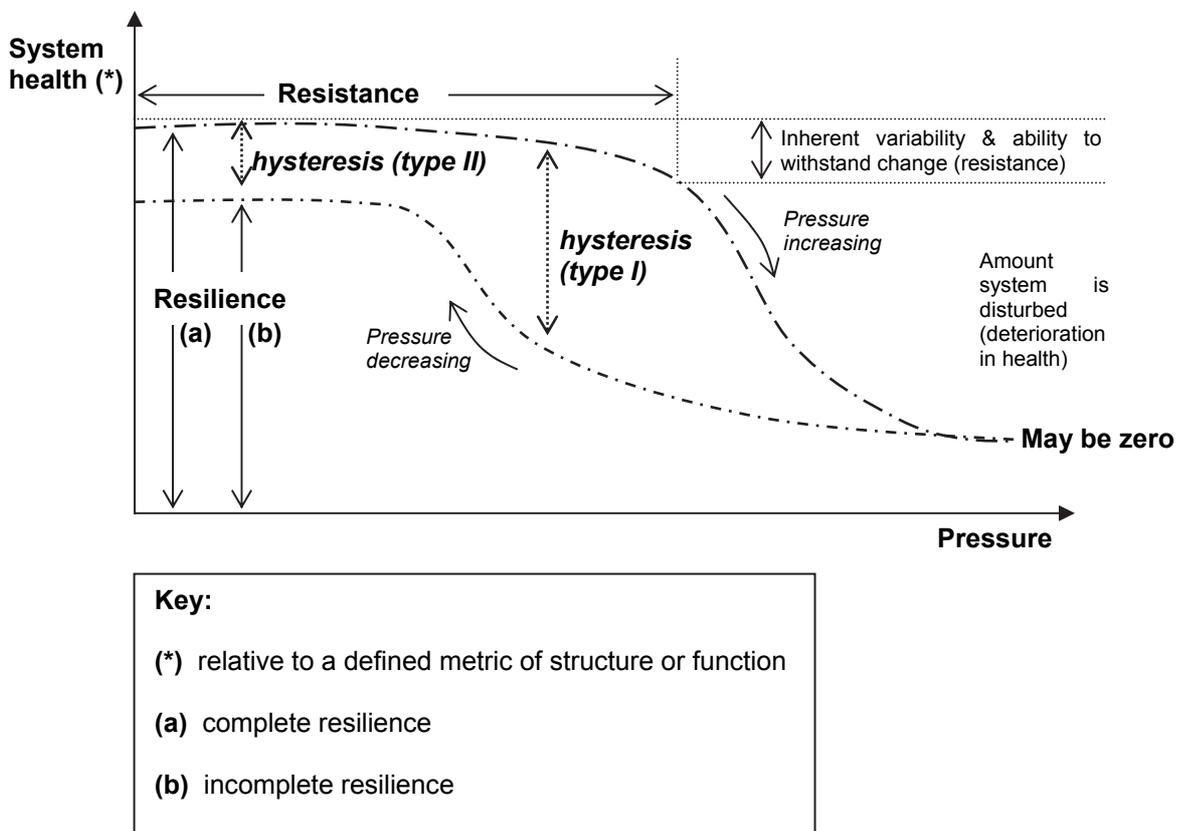


Figure 1. Conceptual model of changes to the state of a system with changing pressure (Elliott *et al.*, 2007)

## 2.2 Variability in the definition of recovery

Some of the most basic definitions of recovery (as trajectory or process) are summarised by Lotze *et al.* (2011) and include:

- **simple increase** – a general increase or improvement in a specified response over time, towards pre-disturbance conditions, over a specified time period or in relation to management measures;
- **standardised or scaled increase** –standardised between species according to (for example) life histories or rates of succession;
- **increase towards a specific target** such as biomass at maximum sustainable yield, maximal population size observed;
- **increase to an historical or pristine level** or to some proportion of former carrying capacity;
- **attainment of former structure or function** such as population structure (adult:juvenile or male:female ratios), trophic levels and interactions (such as predator:prey interactions), functional group (or biological traits) or habitat composition.

However, these definitions indicate a pathway of recovery rather than an endpoint state of recovered. In the context of marine conservation and for the purposes of meeting the requirements of the Marine Protected Areas Project to date, SNH has adopted the definition of recovery (as an endpoint) proposed by MarLIN (2013):

***‘a habitat, biotope or species which has undergone disturbance or damage returns to a state close to that which existed before the activity or event caused the change’.***

A more detailed explanation of this is given in Newell *et al.* (1998) who describe recovery (as a process) as:

***‘the establishment of a successional community which progresses towards a community similar in species composition, population density and biomass to that previously present’.***

Due to natural population variability, whether or not a community can be classed as similar to its pre-disturbance state depends on whether the undisturbed community would have remained stable over time (Newell *et al.*, 1998). In order to account for this, Newell *et al.* (1998) also state that similarity to a ‘characteristic’ non-impacted reference community could constitute recovery. However, this is also problematic due to the difficulties in defining reference conditions, as well as the natural variability within the reference population. Based on this, Newell *et al.* (1998) proposed that a recovered state (as an endpoint) could be defined as:

***‘establishment of a community that is capable of maintaining itself and in which at least 80% of the biodiversity has been restored.’***

This then raises the question of whether biodiversity in this context is defined according to structure (the characteristics at one time such as species composition, structural richness and diversity expressed in the form of an index, abundance and percentage cover etc.) or functioning and the maintenance of a set of rate processes.

Seaman (2007) suggested that full ecosystem restoration (i.e. recovered) can be declared once:

***‘resilience to the normal range of environmental stress and disturbance could be demonstrated together with interaction with surrounding systems in terms of biotic and abiotic flow’.***

Whilst these concepts appear objective and initially readily understandable, defining the point at which a biotope, species or habitat regains its former state is largely subjective and is highly variable (Zajac and Whitlatch, 2003; Lotze *et al.*, 2011; Mazik and Smyth, 2013). This variation predominantly lies in the definition of the ‘previous state’ or reference condition, the choice of parameters or indicators used to compare current and previous states and in the timescale of the study or monitoring programme (Mazik and Smyth, 2013) in relation to the lifespan/turnover of the feature.

### *2.2.1 Establishing a recovery end point*

Borja *et al.* (2012) reviewed the main methods for establishing baselines (or endpoints) for recovery. Baselines were obtained from:

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

according to their value as a methodology for establishing baselines. Due to a long history of human modification of the marine environment, information regarding what that former state (required in most pristine and historical baselines) was may not exist, and return to such a state may be unachievable due to shifting baselines associated with, for example, climate change or natural population variability. Substantial change to the physical environment connected to the period of initial decline may mean that an alternative stable state is favoured by the recovering biological component (Figure 2). Similarly, the return to some pre-human pristine state may be unattainable without removing all human-induced stressors and so aiming for such a reference condition would be unrealistic.

Furthermore, definition of historical reference points is variable according to variation in the methods used in their determination (Kelly and Harwell, 1990; Lotze *et al.*, 2011). In the absence of pristine or historical data, reference conditions can be predicted through the use of modelling techniques. The expected metrics of recovery (as a process or an endpoint) can be modelled based on the characteristics of the habitat or population of interest. Modelling can simulate both the spatial extent of the feature of interest as well as numerical aspects of the population. Modelling outputs containing multiple components, such as a community associated with a habitat, are harder to achieve and the overall confidence may be reduced. To achieve robust model predictions, a large data set that is spatially and temporally intensive is required; in many regions such data sets do not exist (Borja *et al.*, 2012). The use of models without suitable, high quality input information can lead to 1) unrealistic and incorrect predictions, and 2) superficially convincing outputs that confer more confidence about the baseline setting procedure than actually exists. Furthermore, the inherent complexity of some models and the use of required assumptions can also reduce the transparency, objectivity and comprehensibility of the resulting baseline. Based on these issues and inadequacies, Borja *et al.* (2012) considered ‘best professional judgement’ as the best approach for establishing baselines.

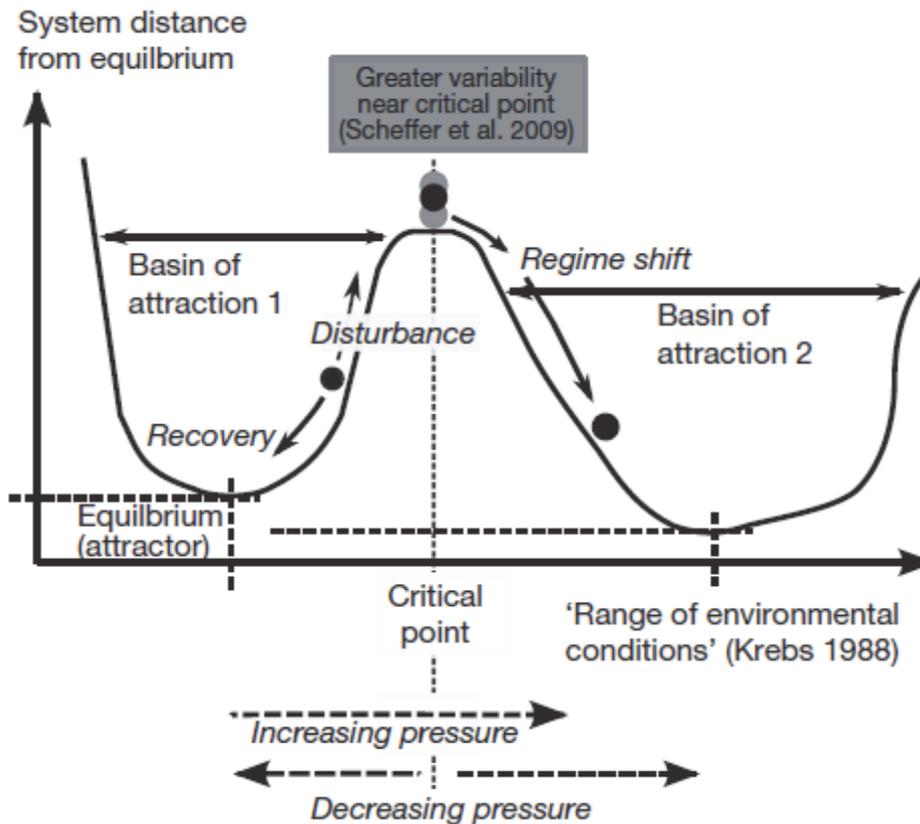


Figure 2. The landscape metaphor for stability and regime shift (Holling, 1973). The ball, representing ecosystem state, moves between valleys denoting different regimes, following disturbance to the ball or changes in the landscape (Figure from Tett et al., 2013).

### 2.2.2 Parameters used to assess recovery and recovered status and study length

There are also difficulties associated with choosing appropriate parameters or indicators to determine recovery (as a trajectory or endpoint) and the choice of indicators is often strongly influenced by the budget available for experimental work or monitoring. Mazik and Smyth (2013) highlighted that parameters used to determine a recovery endpoint ranged from simple measures of species richness, abundance and diversity to that of community structure, population structure and, in a limited number of studies, ecological function. With respect to macrobenthos, most studies refer to species richness (and re-colonisation by the same species present before disturbance), abundance (e.g. Bell and Devlin, 1983; Zajac, 2004; Gilkinson *et al.*, 2005) and evidence of larval recruitment (Gilkinson *et al.*, 2005). Simplistic measures such as this do not account for the relative abundance of the various species (community structure), nor do they account for population structure or ecosystem functioning. A greater number of studies have assessed community structure in some way (e.g. Newell *et al.*, 1998; Dernie *et al.*, 2003; Thrush *et al.*, 2003; Zajac and Whitlatch, 2003; Bolam *et al.*, 2004, Boyd *et al.*, 2005; Herkül, 2011) but very few have assessed population structure (Beukema *et al.*, 1999). In biogenic habitats, measures of distribution and spatial extent of the reef are commonly used and are often associated with measures of density, population structure of the reef forming species, larval settlement, proportion of live and dead material and measures of community structure of the associated community (e.g. Hall-Spencer and Moore, 2000a; Trigg and Moore, 2009). With respect to seagrasses, Hammerstrom *et al.* (2007), Campbell and McKenzie (2004) and Neckles *et al.* (2005) all measured shoot density and spatial extent of seagrass beds and reported recovery times of 2-7 years.

However, the more complex the suite of parameters (in terms of measuring biological complexity), the greater the time period required to achieve recovered status. For example, with respect to seagrasses, this recovery time of 2-7 years increases to >18 years when physiological parameters were considered (recovery of which may be essential to maintaining recovered status in the long-term) (González-Correa *et al.*, 2008). In intertidal benthic habitats, documented times taken to reach a recovery endpoint vary between 6 months and >5 years, based on simplistic measurements of species richness and abundance, and population structure, respectively (Mazik and Smyth, 2013). Thrush *et al.* (2003) also highlighted the importance of sampling strategy in the interpretation of recovery, suggesting that sampling the surface (top 2 cm) and deeper sediment layers (2-15 cm) separately could provide important information on the patterns of colonisation by larvae with high dispersal ability and adults with lower dispersal ability, respectively.

The determination of recovered status also appears to be a function of study length. Indeed, the timescale to achieve a recovery endpoint in many studies has been predicted to be beyond that of the study (generally 6 months to a year) and generally increased with increasing complexity of the parameters being used to determine recovery (Trigg and Moore, 2009; Mazik and Smyth, 2013).

### 2.2.3 Determining convergence between recovering and target communities

Kelly and Harwell (1990) highlighted that the variability of various ecological responses to stress often results in a lack of ability to determine when significant or serious ecosystem effects have occurred. The same applies to the ability to determine whether or not a species or habitat can be considered to have recovered. The current debate regarding the stability of ecosystems also has important implications for how recovery is considered (Tett *et al.*, 2013). The stability debate questions whether ecosystems have a stable equilibrium and that perturbed states move back towards this point, or whether ecosystems are naturally unpredictable and dynamic. If ecosystems have an intrinsic pristine state, then recovery to this point is understandable and predictable. However, if the underlying dynamics of an ecosystem are complex and can exist in multiple states, then the appropriate trajectory for recovery of the ecosystem may be unpredictable, variable and difficult to define (Tett *et al.*, 2013). For example, in an intertidal sediment, Dernie *et al.* (2003) found that natural fluctuations in macrofaunal abundance in both disturbed and ambient sediments gave the impression of convergence between the two communities long before recovery had actually been achieved. This situation may arise as a result of larval settlement at the time of sampling, giving the impression of high abundance (where abundance in the disturbed sediment may exceed that of the ambient habitat or target abundance value), yet high levels of larval and juvenile mortality dictate that a high proportion of settled larvae will not be recruited to the adult population. Hence, abundance values will ultimately decrease, potentially below target values (Newell *et al.*, 1998). Similarly, Thrush *et al.* (2008) and Mazik *et al.* (2010) demonstrated that spikes in settlement over time did not necessarily constitute recovery (as an endpoint), indicating that the duration of the post-disturbance monitoring period was an important factor in understanding recovery.

Newly available habitat may experience an initial influx of high abundance which will eventually decrease through mortality, species interactions and evolution of the physical nature of the habitat (Newell *et al.*, 1998; Mazik *et al.*, 2010) as the habitat and community develop. Similarly, early colonisation by opportunistic species may lead to abundance values which are considerably higher than the reference value. However, these opportunistic species are short-lived and, as the colonisation and recovery processes proceed, will be replaced by lower numbers of other species (Newell *et al.*, 1998) and whilst abundance may decrease, the overall community structure will become closer to the reference or target community. Additionally, Mazik *et al.* (2010) found convergence between biological parameters (species richness, abundance, biomass and overall community structure) in a

newly created habitat and a reference site (target community) after three years of monitoring but that the communities subsequently diverged in later years. This implies that apparent recovery can be short term and monitoring programmes need to be sufficiently long to pick up this temporal variability associated with natural habitat evolution (Mazik and Smyth, 2013). Variability in the reference community has frequently been found to imply recovery in that temporarily low reference values for species abundance, richness and diversity (for example) have been equivalent to those of the recovering community and this has been interpreted as 'recovered' at a single point in time when, in fact, divergence of the two communities may well follow in the future (Mazik *et al.*, 2010; Mazik and Smyth, 2013).

It is highly unlikely that recovery trajectories are linear processes. For example, annual variation in environmental parameters and subsequent modification of biological processes such as recruitment and mortality may result in an oscillating pathway. Density dependent issues may influence the rate of the trajectory towards recovery.

#### 2.2.4 *Lifespan or turnover of the feature of interest*

Borja *et al.* (2010) and Duarte *et al.* (2013) also show evidence that the recovery of an individual ecological component was dependent on the lifespan or turnover time of that component; for example short-lived and fast-turnover species/components such as the meiofauna or plankton would be expected to recover more quickly than long-lived and slow turnover components such as higher plants, birds or fishes.

### 2.3 **Practical definitions of species and habitat 'recovery' and 'recovered' and issues when assessing recovery**

Any definition of recovery should account for the difference between '**recovery**' as a **process** and '**recovered**' as an **end point**. 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).

From a practical point of view, a workable definition of recovery and recovered 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. This may result in a degree of compromise between ecological integrity and economic realities of the required monitoring. For example, a definition of recovery detailing population structure (age class, size class) may be unworkable if these parameters have only been measured in a single study and if it is not viable in terms of time and money to carry out these measurements for all interest features in all areas.

Aspects of recovery also include spatial extent and, for biogenic species, reef or nest integrity and stability. It may be that a particular biological component has been lost or heavily depleted in certain areas, and recovery is dependent on recolonisation of these areas. The same difficulties of establishing a baseline with non-spatial components of recovery also apply to spatial targets. One must have prior or predicted information on the pristine or historical distribution of a species or habitat. As with other metrics of recovery, this information is typically not present. Habitat suitability models can provide some of this information but are typically based on small amounts of existing ground truthing data and models using subjective assumptions. There is also a practical issue of monitoring enough spatial sites to establish the long-term recovery of the feature of interest. The cost

implications may lead to a more measurable target rather than one with strict ecological thresholds.

As previously stated, natural spatial and temporal variability in both a reference state and recovering status makes it difficult to set strict numerical targets for population density, community structure and spatial extent. Therefore, the **recovered reference** 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. This is referred to in the following sections as the 'recovered reference range'. This may mean collating data from the best known examples to form assumptions about 'typicalness' for that particular feature in terms of (for example) density, patchiness, stability and the nature of associated community. In terms of a specified set of parameters, 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.

Based on the discussion above, the following practical definitions of recovery and recovered are suggested below. The definitions should be applied on a site-by-site basis.

### 2.3.1 *Over-arching definitions*

The required elements of 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) measureable/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; nest/reef integrity as well as spatial extent);
- 6) representative of the required legislative end-point (e.g. favourable conservation status);
- 7) self-sustaining (i.e. a self-sustaining based on interactions between component sub-populations<sup>1</sup> of a meta population<sup>2</sup>);
- 8) based on the best available information.

Approaches to establishing the Recovered Reference Range include:

- 1) observations of pristine conditions;
- 2) historical observations of the feature at the location;
- 3) modelled predictions; and
- 4) best professional judgement of parameters and spatial extent.

These sources are listed in order (high to low) of desirability, data requirements, objectivity and confidence/certainty.

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<sup>1</sup> 'Sub-population' refers to a small population unit that is connected via physiological preferences to other population units.

<sup>2</sup> 'Meta-population' refers to a collection of sub-populations.

A feature can be considered to have **recovered** when the parameters or indicators being used to measure recovery are maintained within a defined Recovered Reference Range, and **recovering** whilst moving towards this reference.

Therefore, **Recovery** is considered as:

- *a consistent trajectory, detectable above natural variability, of a representative set of feature properties from a previous (or otherwise defined) state towards the 'Recovered Reference Range', throughout a spatially explicit area.*

**Recovered** is considered to be a:

- *stable, long-term similarity of a representative set of feature properties of a recovering feature to the Recovered Reference Range throughout a spatially explicit area.*

### 2.3.2 Working definitions

#### Single species

**Recovery/recovering** - 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.

**Recovered** - 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)

**Recovery/recovering** - 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.

**Recovered** - 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.

Biogenic habitats require both single species and community definitions of recovery and recovered.

#### Physical habitats

**Recovery/recovering** - 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.

**Recovered** - 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.

### 2.3.3 *Monitoring requirements*

Due to the likely timescales required to reach a recovered endpoint (for many of the species considered in this document), there is a potential need for both 'recovery' monitoring and 'end-point/recovered' monitoring. There may be benefits to separating the two objectives. Monitoring of the recovery process is necessary to understand the recovery trajectory and to provide an indication that management actions are beneficial and worthwhile. Similarly, it will confirm no further decline. This requires spatial and temporal components.

Monitoring of 'recovered status' (as an end point) will ensure that a degree of stability is achieved and that the recovered population, community or habitat remains in a state where it can be considered sustainable. That is, species composition, density, population structure, establishment of associated community etc are stable over time (with no long term pattern of decline), accounting for natural variability. Ideally, the objective would be to achieve statistical similarity although where variability is high, this approach is unlikely to be valid and expert judgement would be required. End-point monitoring requires both spatial and temporal components.

It is emphasised that the monitoring strategy should be developed on a feature-specific basis.

#### **Box 1. Conclusions regarding recovery concepts**

- Definitions are provided to separate the recovery process from the recovered end point.
- The use of a 'recovered reference range' is recommended to provide a clear, quantitative recovery target. The information required for the recovered reference range is described for species, species assemblages and physical habitats.

### 3. CURRENT KNOWLEDGE OF RECOVERY IN TEMPERATE MARINE ECOSYSTEMS

In order to develop a framework for assessing recovery potential, it is necessary to examine the evidence for recovery for the selected features of interest (i.e. all of the high priority features and half of the medium priority features contained in Table 1). This section takes each feature in turn and identifies the nature of the disturbance (making the distinction between controlled, experimental disturbance and that associated with anthropogenic activities) and, where recovery has been documented, the patterns and timescales of that recovery. Where recovery has been partial, slow or has not been documented, species or habitat-specific processes which may inhibit recovery have been identified. The factors influencing recovery are further discussed more generically, using examples from other habitats, in section 4.1.

Details of the habitat requirements and distribution in Scottish waters of the features reviewed in this section are given in Lancaster *et al.* (2014). The Feature Sensitivity Assessment Tool (FEAST) (Scottish Government, 2013) was used to identify the activities and pressures to which each species or community is sensitive. This provides a generic indication of the links between activities, pressures and the sensitivity of each feature. It should be noted that the level of confidence associated with many of the sensitivity assessments is low.

#### 3.1 *Limaria hians* (flame shell)

The semi-infaunal bivalve *Limaria hians* has been described as a keystone species due to its tendency to form biogenic reefs (formed by the joining of nests of multiple animals) that may cover hectares of sea bed, standing 10-20 cm high, and contain a highly diverse community of associated and dependent conspecifics (Hall-Spencer and Moore, 2000a). Around 284 species have been found to be associated with *L. hians* beds, including a variety of encrusting organisms, the echinoderms *Ophiothrix fragilis*, *Ophiocomina nigra* and *Asterias rubens*, the gastropod *Buccinum undatum*, the anthozoan *Alcyonium digitatum* and the hydrozoans *Plumularia setacea*, *Kirchenpaueria pinnata* and *Nemertesia* spp. Decapod crustaceans are also usually present. Infaunally, a variety of polychaetes and bivalves have been found (Hall-Spencer and Moore, 2000a).

*Limaria hians* beds form a specific biotope (*L. hians* beds in tide-swept sublittoral muddy mixed sediment SS.SSMx.IMx.Lim) found in sheltered areas, such as sea lochs, of mixed sediment with variable current speeds (>1-6 kn) and a depth range of 5-30 m (Connor *et al.*, 2004). *L. hians* typically inhabits the shallow sublittoral (maximum 100 m depth) in weak to strong tidal streams (Minchin, 1995; Hall-Spencer and Moore, 2000a). It is associated with variety of coarse sediment types including coarse sands, gravels, mixed muddy gravel and maerl (Hall-Spencer and Moore, 2000a; Connor *et al.*, 2004) and may be found amongst kelp holdfasts (Minchin, 1995; Trigg and Moore, 2009). There is a degree of overlap in the habitat requirements of *L. hians* and *Modiolus* and the two species may coexist and may also be found in association with maerl habitats (Hall-Spencer and Moore, 2000a).

##### 3.1.1 Threats and sensitivity

*Limaria hians* is highly sensitive to physical disturbance, particularly in the form of abrasion and habitat removal/change to another habitat type (Hall-Spencer and Moore, 2000a; Tyler-Walters, 2003). Similarly it is highly sensitive to extraction, usually non-selective extraction associated with scallop dredging. Changes in current speed may modify the sea bed in terms of erosion and increased sediment mobility or deposition of fine material, and *L. hians* is thought to have medium sensitivity to this. The species is highly sensitive to contamination (e.g. Minchin, 1995) but sensitivity to specific groups of compounds, nutrient enrichment and low dissolved oxygen is not well understood (Tyler-Walters, 2003).

In the context of Scottish waters (based on FEAST, Scottish Government, 2013), fishing activities (including hydraulic dredging, otter trawling and scallop dredging) are of greatest relevance to *L. hians* as a result of these activities resulting in physical damage (abrasion), long term physical modification of the seabed (physical change to another habitat type) and direct removal, damage or mortality of *L. hians*. *L. hians* is less sensitive to other fishing activities (creels and pots, demersal seine netting, dive fisheries and fixed nets) in that there is some potential for these activities to be carried out in areas where *L. hians* is present as the damage may be patchy and significantly lower than that associated with, for example, scallop dredging. Aquaculture (shellfish and finfish) is a potentially damaging activity where *L. hians* beds are in close proximity to farms due to increased organic input, smothering of the bed and low dissolved oxygen.

Where complete defaunation has not occurred, sensitivity to changes in current speed and sediment mobility are likely to increase following disturbance as a result of reduced nest integrity and nest fragmentation, which may enable increased erosion, displacement and dispersal of individual *L. hians*, coupled with uprooting by displaced *Laminaria* sp. and other large seaweed species attached to the nest (Minchin, 1995; Trigg and Moore, 2009). Vulnerability to predators will also be increased (Hall-Spencer and Moore, 2000a).

### 3.1.2 Overview of relevant studies

Studies on the impacts of disturbance (particularly fishing disturbance associated with bottom towed fishing gear) are relatively common but few assess the recovery times of *L. hians*. Therefore it is not possible to assess differences in recovery potential following different spatial scales of disturbance. However, two studies representing two different pressure types indicate that *L. hians* can recover following disturbance. Trigg and Moore (2009) experimentally simulated the impacts of scallop dredging (representing physical abrasion of the substratum) on *L. hians* beds by removing a number of nest areas of 0.25 m<sup>2</sup> and raking the underlying sediment to disturb the physical structure and remove large stones (as a Newhaven dredge would do). Twelve months after the disturbance, mean nest coverage in the disturbed plots was less than 25%, with a maximum value of 36%. Therefore, whilst recovery was not complete (in terms of equivalent nest coverage before and after disturbance), the process had started. It is of note that recovery of nest material was predominantly via encroachment from the sides of the cleared areas (although some growth did occur in the centre) and was therefore largely reliant on the existence of intact individuals. Depending on the nature of the physical damage, there may be scope for recruitment and bed recovery via larval dispersal (Minchin, 1995), assuming that a source of larvae exists.

In an experiment such as that carried out by Trigg and Moore (2009), the disturbance was patchy and disturbed plots were surrounded by undamaged nest and control plots of 100% cover in close proximity. Such a mosaic effect enables migration of *L. hians* between plots. In contrast, scallop dredgers disturb much larger areas of the bed and a single pass of the gear can almost completely remove a reef leaving only damaged individuals which are rapidly consumed by scavengers (Hall-Spencer and Moore, 2000a). Furthermore, destabilisation of the sediment can result in burial of surviving individuals, increasing overall mortality (Minchin, 1995). A scallop fishing boat would typically have up to 14 Newhaven dredges attached (seven on each side), depending on the size of the vessel (Galbraith *et al.*, 2004; O'Neill *et al.*, 2013), each with a width of 82 cm and a set of 9 cm teeth spaced approximately 7 cm apart (O'Neil *et al.*, 2013). This potentially equates to a dredge track of up to 11.5 m wide, of variable length. Clearly, this area of disturbance is significantly larger than that associated with the experimental work carried out by Trigg and Moore (2009). Given that the major route to recovery was encroachment from the sides of the disturbed area and that the majority of the remaining *L. hians* are likely to have been in good condition (due to the careful way in which nest material was removed from the disturbed plots), this

implies long, and indeed uncertain, recovery times for reefs damaged or removed following scallop dredging or other activities causing a similar degree and spatial extent of impact. Indeed, for a dredged area of 7.5 m wide, Trigg and Moore (2009) estimated a recovery time of 117 years, assuming linear growth of 3.2 cm per annum but without considering any other influences on nest growth. Clearly this recovery time would increase with increasing dredge size, such as that described by O'Neill *et al.* (2013). Furthermore, nest coverage cannot necessarily be equated to recovery since newly built nests were much thinner than older, established nests (Trigg and Moore, 2009) and they may therefore be more susceptible to disturbance such as that associated with uprooting of kelp holdfasts or stormy conditions.

The second study (Minchin, 1995) assessed recovery of *L. hians* following population decline resulting from exposure to TBT in the Morross Channel, Northern Ireland. Five years of settlement failure, coupled with an absence of larvae in the water column, followed intense settlement in 1982 and population decline to 1.6% of the abundance recorded in 1980, were attributed to TBT contamination. Following a ban on the use of TBT, larval settlement was observed in 1986 and 1987 and by 1994, densities of *L. hians* appeared to have recovered to their 1980 state as indicated by 100% coverage (no extensive patches of bare sediment) and the reappearance of an extensive byssal carpet. Laminarian seaweeds had also recolonised. Most importantly, *L. hians* is capable of reproducing in its second year so assuming successful recruitment following the 1986/1987 settlement events, the presence of a spawning population would have increased the potential for recovery. The total time required to reach a recovered state in this study was nine years.

Neither study assessed the recovery of the infaunal and epifaunal communities associated with *L. hians* reefs. In particular, Minchin (1995) and Trigg and Moore (2009) both highlight the dependence of *Laminaria* spp. and *Sacchoriza polychides* on *L. hians*. Considering that kelp and seaweed communities on sublittoral sediment (SS.SMp.KSwSS) are a feature of conservation interest, and the fact that the ability of biogenic habitats to support a diverse community is one of the main components of their ecological importance, this knowledge gap is of relevance.

### 3.1.3 Factors influencing recovery

#### 3.1.3.1 Size of disturbed area and nature of disturbance

Trigg and Moore (2009) suggested that recovery potential was related to the nature of the disturbance and that severe physical disturbance, such as that associated with scallop dredging, would be associated with longer recovery times than disturbance from chemical contamination, where the physical structure of the sediment is not affected. Indeed, Minchin (1995) highlighted the susceptibility of *L. hians* to burial in destabilised sediments. However, this may not be the case for deposit feeders or organisms without shells (associated with *L. hians* beds) which would be more susceptible to uptake of contaminants through direct contact with the sediment and interstitial waters.

#### 3.1.3.2 Density of remaining individuals

The recovery patterns observed by Trigg and Moore (2009) were facilitated by the patchy distribution of the disturbance within an intact *L. hians* bed with 100% cover of *L. hians* individuals. Colonisation was via lateral expansion from the perimeter of the disturbed area. Where large scale disturbance has taken place, the density of the remaining individuals is low and there is a large distance between intact individuals or nests, recovery times are likely to be prolonged.

### 3.1.3.3 Pressure removal

Both Minchin (1995) and Trigg and Moore (2009) demonstrated that recovery of *L. hians* is possible following complete removal of the pressure, assuming that there is a source of adult colonists or larval recruits.

### 3.1.3.4 Swimming ability and dispersal mechanisms

*L. hians* has the ability to swim (Hall-Spencer and Moore, 2000a) indicating the potential for dispersal, redistribution and settlement following disturbance, provided that suitable habitat can be found. Donovan *et al.* (2004) found that *Limaria fragilis* could swim at speeds of up to 7 cm s<sup>-1</sup> (body length of 40 mm) and that swimming speed increased with body length. However, whilst no information on swimming distance was found, the mechanism by which *Limaria* propels itself through the water is likely to be energetically expensive and dispersal over large distances through swimming activity is not expected. Therefore, re-establishment of *L. hians* populations is likely to be limited to areas where suitable habitat is present within close proximity to a source of colonists.

Fecundity for *L. hians* is assumed to be high (as for other bivalve species (Gosling, 2003) and, since larvae are released directly to the water column, dispersal potential is expected to be high, depending upon the hydrodynamic conditions. Spawning is thought to occur between July and September, although further south (Adriatic Sea), *L. hians* is sexually active for most of the year (Trigg, 2009). This indicates that recovery associated with larval settlement and recruitment may be limited to a few months of the year. Trigg and Moore (2009) found little evidence for recovery as a result of larval settlement and recruitment although Trigg (2009) states that this may be a function of the experimental design. In contrast, Minchin (1995) was able to attribute recovery to larval settlement events and noted maximum densities of larvae in the water column in late summer, indicating greatest potential for the recovery process to begin during this period.

### 3.1.3.5 Susceptibility to predation

Due to its inability to close its shell, *L. hians* has developed defensive adaptations whereby it secretes a viscous mucus to deter predators (Gilmour, 1967). Where reef damage has occurred, this suggests that any remaining surviving individuals could potentially re-settle as described in section 3.1.3.4. Intact individuals would be able to deter predators via the mechanism described by Gilmour (1967). However, Hall-Spencer and Moore (2000a) noted that the shells of *L. hians* are extremely delicate and that following mechanical disturbance (such as that associated with towed fishing gear), any damaged surviving individuals would be consumed by predators such as cod, dragonets, edible whelks, brittlestars and decapods crustaceans, within 24 hours. This indicates that damaged *L. hians* lose their ability to swim and deter predators.

### 3.1.3.6 Timing of recovery

Trigg and Moore (2009) did not find any seasonal difference in the rate of nest re-growth indicating that the time at which a pressure is removed and the recovery process is allowed to begin is not an important factor in determining the recovery potential of *L. hians*. However, this study was only carried out over a 12 month period (with 2 surveys at 6 monthly intervals) and therefore does not account for inter-annual variability. Furthermore, larval settlement was not assessed yet Minchin (1995) identified this as a mechanism of recovery and larval settlement is dependent upon spawning patterns. If a pressure is removed immediately before a spawning period (late summer, generally July-September), recovery is likely to proceed more quickly than if it were removed after a recruitment event, whereby new recruits may be damaged or removed several months before the next spawning event (Beukema *et al.*, 1999).

## **Box 2. Conclusions regarding the recovery potential of *Limaria hians***

Whilst studies on the recovery of *L. hians* are limited, the two that were found both indicate that, following pressure removal, this species can recover, assuming suitable substratum conditions. However, the nature of the disturbance in these studies was different (chemical versus physical disturbance). Following contamination by TBT, Minchin (1995) indicated that recovery may take less than 10 years for demographic recovery – no information was available for the recovery of the associated bed assemblages. In contrast, Trigg and Moore (2009) indicated that following physical disturbance associated with scallop dredging, recovery may take over 100 years to achieve spatial coverage but the time for recovery of nest thickness (a feature that may influence nest stability) is not known. It is also emphasised that these estimates of recovery time assume linear recovery in the absence of other influences.

### **3.2 *Modiolus modiolus* (horse mussel)**

*Modiolus modiolus* is one of the larger, long-lived mytilid mussels. Individuals aggregate into biogenic reef structures and, as such, this species is regarded as a keystone or foundation species for four habitat types currently recognized in Europe (Connor *et al.*, 2004; Rees 2009):

1. *Modiolus modiolus* beds with hydroids and red seaweeds on tide-swept circalittoral mixed substrata (EUNIS Code: A5.621; JNCC 04.05 code: SS.SBR.SMus.ModT);
2. *Modiolus modiolus* beds on open coast circalittoral mixed sediment (EUNIS Code:A5.622; JNCC 04.05 code: SS.SBR.SMus.ModMx);
3. *Modiolus modiolus* beds with fine hydroids and large solitary ascidians on very sheltered circalittoral mixed substrata (EUNIS Code: A5.623; JNCC 04.05 code: SS.SBR.SMus.ModHAs);
4. *Modiolus modiolus* beds with *Chlamys varia*, sponges, hydroids and bryozoans on slightly tide-swept very sheltered circalittoral mixed substrata (EUNIS Code: A5.624; JNCC 04.05 code: SS.SBR.SMus.ModCvar).

*Modiolus modiolus* is found intertidally and to depths of 280 m subtidally (Tyler-Walters, 2007). This species is found both infaunally and epifaunally (although typically described as semi-infaunal) in coarse, high energy environments and sheltered, soft sediment habitats. *Modiolus modiolus* forms aggregated reefs as it develops (Lindenbaum *et al.*, 2008). In soft sediment habitats, these aggregations are small and form distinct clumps (Maggorian and Service, 1998) whilst in higher energy environments, these aggregations can form large ridged biogenic reefs or beds, orientated perpendicular to the current. These beds can reach a thickness of 1 metre on top of the underlying coarse sediment. In sheltered, soft sediment environments, the presence of *M. modiolus* beds provides hard substrata (live and dead shell) for a diverse and abundant epifaunal community (Maggorian and Service, 1998). *M. modiolus* beds are able to remove and store large amounts of organic matter in the form of body tissue. They are also capable of high filtration rates that enable the concentration of large amounts of suspended particulates, from pelagic waters, into energy-rich faeces or pseudo-faeces, which can be utilized as food by other species (Navarro and Thompson, 1997).

#### **3.2.1 Threats and sensitivity**

The principle sensitivities of *Modiolus modiolus* beds, according to FEAST (Scottish Government, 2013), are surface and sub-surface abrasion/penetration associated with

activities such as aggregate extraction and fishing (e.g. hydraulic dredging, scallop dredging and otter trawling), siltation associated with these activities, and the removal of non-target, target species. Organic enrichment and siltation rate changes, associated with fin and shellfish aquaculture, are moderately tolerated by *M. modiolus*. *M. modiolus* is moderately sensitive to temperature and hydrodynamic changes as well as local modification of the wave exposure regime (FEAST, Scottish Government, 2013).

Survey work undertaken by Magorrian and Service (1998) in Strangford Lough indicated that trawling activity resulted in the removal of large epifaunal species on the biogenic reef, such as *Alcyonium digitatum*, and the physical scattering of the *M. modiolus* clumps to give an overall flattened appearance to the beds. There was also a substantial king scallop fishery within Strangford Lough, using scallop dredges, during the same period (Roberts *et al.*, 2004). Trawling and dredging consequently reduced the habitat complexity of the *M. modiolus* communities and thereby degraded the hard substratum used for epifaunal attachment and specific microhabitats within the structure of the mussel beds used by interstitial species.

### 3.2.2 Overview of relevant studies

There are few studies documenting the long-term change in beds of *Modiolus modiolus* following disturbance. Monitoring of aggregate extraction sites where *M. modiolus* is found has provided several useful sources of information on recovery, for example, Kenny and Rees (1996). The well documented decline of *M. modiolus* in Strangford Lough and the recent commissioning of the *Modiolus* Restoration Research Project at Queen's University, Belfast, also provides a valuable case study for understanding the potential for recovery of this species as multiple restoration techniques have been trialled during the project.

Kenny and Rees (1996) observed a significant shift in benthic community composition, containing *M. modiolus*, following a period of marine aggregate extraction. *M. modiolus* was present before extraction commenced but failed to recolonise during the two year period following the event. The resulting community was dominated by a few r-selected species, such as *Dendrodoa* and *Balanus* that were able to recolonize quickly and tolerate the increased amounts of silt transported from the destabilised substratum and overall hardening of the substratum. Recolonisation of *M. modiolus* after the 2 year observation period has not been reported. Due to the slow growth and longevity of *M. modiolus*, studies indicate that recovery of the mussel population, reef structure or associated biodiversity has not been achieved. Although some juveniles have been found within damaged beds of *M. modiolus* in Strangford Lough, Northern Ireland (Dai Roberts, Queen's University, Belfast pers. com.) this does not constitute recovery. To date, there has been no documented recovery of the *M. modiolus* beds, or more importantly, the biogenic structure that provides habitat for other species and weight of individuals required to modify ecosystem processes. Individual components of recovery have been investigated and are included in the relevant sections below.

### 3.2.3 Factors influencing recovery

#### 3.2.3.1 Spawning and recruitment

The frequency, regularity and magnitude of spawning is highly variable and changes significantly with temperature, habitat and depth. In sheltered, soft sediment habitats within Strangford Lough, Northern Ireland, *M. modiolus* has been observed to be a trickle-spawner (slow, continuous release of gametes over longer periods) (Seed and Brown, 1975; Brown and Seed, 1977). However, Swedish and northern Norwegian populations showed a distinct spawning period in June - July respectively (Brown, 1984). The regularity of spawning between years also differs greatly. Most populations in British waters spawn annually,

whereas Wiborg (1946 in Brown, 1984) reported that spawning occurring only every 2nd to 3rd year in Norwegian waters.

Recruitment success in populations of *M. modiolus* is generally reported to be very low, even when compared to levels of bivalve recruitment generally. Regular recruitment was observed in populations in Strangford Lough (Seed and Brown, 1978). However, the recent decline of *M. modiolus* in Strangford Lough, suggested to be caused by fishing activity and specifically scallop dredging, has also been associated with a decline in recruitment levels (Elsäßer *et al.*, 2013). Very irregular recruitment, with gaps of many years was also reported for Norwegian (Wiborg, 1946 in Brown, 1984) and Canadian populations (Rowell, 1967 in De Schweinitz and Lutz, 1976). Scottish populations varied, with 'normal' recruitment occurring in areas of strong currents, resulting in a relatively young population, while recruitment was negligible in areas of quiet water resulting in an ageing population, and in a deep water population no recruitment had occurred for a number of years and the population was old, possibly senile and declining (Comely, 1978). The observed difference in recruitment between sites is suggested to be a product of varying habitat suitability (poorer habitat, lower growth and higher mortality) and connectivity between sub-populations driven by hydrodynamics (less larvae and settlement at the sites).

### 3.2.3.2 Small brood stock size in isolated populations

Although regionally abundant within the British Isles, colonisation by *M. modiolus* of sea lochs has led to the development of discrete sub-populations. Hydrodynamic modelling by Elsäßer *et al.* (2013) suggested that this type of population is likely to be isolated from other populations of *M. modiolus* and therefore dependent on the local sub-population for sustaining recruitment. Elsäßer *et al.* (2013) proposed that larval dispersal capability is highly dependent on the local hydrodynamics and that the flow of larvae between populations in differing water bodies may not be as great as would be expected. Within Strangford Lough, Northern Ireland, hydrodynamic modelling by Elsäßer *et al.* (2013) suggested that hydrodynamic forcing of the dispersal may even prevent dispersal between *M. modiolus* reefs within this water body. Healthy, isolated sub-populations appear to have sufficient brood-stock to be self-sustaining. However, should the adult population be decreased significantly, as has happened in Strangford Lough (Northern Ireland), cumulative density-dependent Allee effects (Allee, 1931), i.e. decreased gamete fertilisation and increased mortality (related to the reduction in adult reef preferred for settlement and juvenile growth), have been observed which are related to a small, isolated brood stock size (Fariñas-Franco *et al.*, 2013).

In isolated or fragmented populations, the residual brood stock is the sole source of gametes for external fertilisation and ultimately recruitment. This situation increases the probability of Allee effects becoming apparent in the population. Sufficient gametes must be released to overcome water column dilution and maintain external fertilisation rates. Furthermore, the supply of larvae must be sufficient to overcome the mortality suffered by larvae and juveniles. Impairment at any point constitutes an Allee effect (Kramer *et al.*, 2009).

Another issue regarding the adult population size is settlement success. The gregarious behaviour of *M. modiolus* is thought to significantly increase the survival of the early life stages. In a recent experiment using cultch (dead shell) in Strangford Lough, Fariñas-Franco *et al.* (2013) found no *M. modiolus* recruitment on cultch deployed on trays after more than one year on the sea bed. However, *M. modiolus* spat recruitment was present amongst live *M. modiolus* translocated for the experiment. As with other bivalve reefs, gregarious settlement behaviour and increased juvenile recruitment within adult conspecifics can generate a negative feedback that hampers restoration efforts. Should the adult population be reduced to a point where restoration is required, the population has to overcome both the problems of reduced reproductive output and diminished habitat (the adult population)

preferred for settlement and successful recruitment in the adult population (Mann and Powell, 2007).

#### 3.2.3.3 Low fertilisation rates

Although low fertilisation rates are partially a product of small brood stocks, specific elements of the life history of *M. modiolus* in sheltered habitats aggravate this situation further. Allee effects in a population of *M. modiolus* in Strangford Lough, Northern Ireland, are an accumulation of several density-dependent factors (Elsäßer *et al.*, 2013). Within this sheltered sea lough environment, *M. modiolus* shows less pronounced reproductive bouts and functionally becomes a trickle spawner (Seed and Brown, 1975). At larger population levels within enclosed water bodies this is unlikely to be a detrimental strategy. However, should the brood-stock become depleted or isolated, this will lead to a greater amount of dilution as gametes are not concentrated in temporally-discrete reproductive bouts. This dilution will, in turn, lead to lower fertilisation rates and thereby generate an Allee effect that ultimately reduces the potential for recruitment. Within Norwegian populations of *M. modiolus*, although spawning bouts within years were much more pronounced, spawning between years was infrequent (Wiborg 1946 in Brown, 1984), suggesting that recruitment can be naturally variable between years naturally.

#### 3.2.3.4 Variable dispersal success

The duration of the planktonic phases is highly variable between studies. De Scheinitz and Lutz (1976) found that larvae collected from Chamberlain, Maine remained in suspension for approximately 19 days before settling. Fariñas-Franco and Roberts (2014) found that the minimum planktonic phase for *M. modiolus* larvae in Strangford Lough is 34 days with most larvae settling after 56 days. This is a significant amount of time spent in the planktonic phase. However, the dispersal of this larvae is ultimately dependent on the hydrodynamic conditions experienced at the spawning site. Comely (1978) observed very low rates of settlement in *M. modiolus* in Scottish waters and attributed much of this low survival to hydrodynamic control of larval supply at the three study sites. At one site, residence time was high and therefore the dispersal of larvae was concentrated locally with no apparent import of larvae from other *M. modiolus* populations. The two other sites Comely (1978) examined were subject to significant along shore currents that swept much of the larvae into the open waters of the Firth of Clyde where they were lost to unfavourable conditions.

Hydrodynamic modelling by Elsäßer *et al.* (2013) also indicated limited dispersal in *M. modiolus* population within Strangford Lough, Northern Ireland. The model firstly indicated that particle exchange between the Lough and the Irish Sea was minimal suggesting that sub-populations of *M. modiolus* in sheltered or enclosed water bodies may often be isolated and often self-sustaining. It was also stated that simulated larval releases within the Lough resulted in a negligible transfer of particles in an east–west/west–east direction across the main tidal flow, indicating low connectivity between the remaining reefs. Dispersal of larvae appears to be strongly related to the prevailing hydrodynamics within the habitat of the adult population. High energy, open environments may be reliant on other source populations for larvae. By contrast, those sheltered water bodies associated with high residence times are likely to be self-supporting.

#### 3.2.3.5 High juvenile mortality

Juvenile mortality is a particularly important factor in influencing the growth and sustainability populations of many marine organisms. *Modiolus modiolus* is a long-lived species with many populations having a large proportion of individuals that are between 35 and 50 years old (Anwar *et al.*, 1990). From analyses of surface growth rings and measurements of mussels grown in experimental cages, Seed and Brown (1978) concluded that the post-settlement growth strategy was heavily influenced by the intensity of early stage mortality. Within in

Strangford Lough, Northern Ireland, Seed and Brown (1978) found the population was contained within two modal groups. The larger of the two modal groups included mussels from around six to over 20 years of age. The smaller modal group which consisted mainly of mussels less than four years old, which represented a balance between slow but continuous recruitment, rapid growth and heavy mortality. The bimodal structure of this population was explained in terms of the life history characteristics of *M. modiolus*. Mussels did not achieve sexual maturity until they reached about 35-40 mm in shell length by which time they were between four and six years old. However, these small but fast-growing mussels were selectively consumed by a variety of benthic predators particularly crabs and starfish. Only when the juvenile mussels had escaped the intense initial predation by effectively growing beyond the size range most vulnerable to attack was energy redirected towards reproductive development. The second size class indicates a reduction in somatic growth but substantial increase in longevity (Seed and Brown, 1978). Ultimately, the influence of juvenile mortality is sufficient to be generating the two distinct modal classes, which also dictates the overall size distribution of the entire population within Strangford Lough.

As *M. modiolus* prefers to settle amongst conspecifics, clumps of adults typically harbour high densities of spat fall densities (Rees *et al.*, 2008; Roberts, 1975). The complex network of micro-crevices created by the matrix of byssus threads and shell fragment found within the clumps of live *M. modiolus* probably provides the juveniles with a significant refuge from predators. Once again, with a diminished adult population, the availability of high quality, adult *M. modiolus* refugia is also reduced which may increase the overall level of juvenile recruitment.

#### 3.2.3.6 Slow adult growth and establishment of a colonised biogenic structure

Growth rates slow appreciably after sexual maturation that many populations do not have distinct size classes for individual broad year cohorts, rather the sexually mature proportion of the adult population coalesce into one adult size class. Adult growth rates vary from 3.5 mm per year (Jasim and Brand, 1989) to a maximum of 8.75 mm (Anwar *et al.*, 1990). The growth rate may be higher in certain populations in faster flowing water, e.g. individuals on oil rigs in the North Sea (mussels on the legs experience increase water flow when outside the benthic boundary layer (Holt *et al.*, 1998)). Comely (1978) also reported that a deep water population (200 m) in Scottish waters had reduced growth rates, possibly due to reduced food availability.

There are two implications of slow growth for restoration of *M. modiolus* biogenic reef. First, body size is typically related to higher reproductive output, hence smaller individuals contribute less larvae than larger mussels. Secondly, *M. modiolus* is in itself a fairly common species; its conservation value lies in the formation of biogenic reef which provides a hard and varied substrata for additional infaunal and epifaunal species. The biogenic reef structure, whether it is the ridge formation in higher energy environments or the smaller clumped outcrop structure in sheltered environments, is a product of accumulation of adults within a mature biogenic structure. Equally, in the soft sediment habitats, significant amounts of dead shell provide clumps of live *M. modiolus* elevation over the substratum and additional volume for settlement. Slow adult growth rates increases the time required for biogenic reef recovery, regardless of the overall recruitment into the population.

A new biogenic reef structure may be considered to have only reached a recovered stage when it is fully colonised by the associated resident biodiversity. The effects of the horse mussel on other organisms is likely to be related to its biological activity, mainly biodeposition (Navarro and Thompson, 1997), and/or the provision of physical habitat (Sanderson *et al.*, 2008). With regard to the epifaunal community, Fariñas-Franco *et al.* (2013) states that the attraction of epifauna towards horse mussel clumps is a result of their three-dimensional configuration influencing small-scale local hydrodynamic patterns. As

individual mussels orient themselves in different directions, turbulence increases particular suspension providing ideal feeding conditions for sessile epifauna. This influence is likely to be greater when mussels are clumped and in complex biogenic structures. This structure is a product of older adult mussel accumulations and is again limited by the slow growth of *M. modiolus*.

Ragnarsson and Burgos (2012) investigated the relationship of three epifaunal species, physical reef structure and the underlying sediments. Species richness was found to be greatest in the coarsest habitat and the areas containing the highest abundance of *M. modiolus*. While the effects of these two factors were synergistic, analysis revealed that *M. modiolus* abundance had a greater influence on the attraction of common epifaunal species. Accumulation of epifaunal species was also suggested to be related to patch diameter. The relationships between the abundance of the dominant epifaunal taxa and species richness with *M. modiolus* abundance were characterised by a sharp initial increase that levelled off after moderate densities of *M. modiolus* abundance was reached.

There is also a significant infaunal diversity associated with *M. modiolus* biogenic reef. The substantial amounts of faeces and pseudofaeces presumably attracts these species to the reef. This is again a feature of mature reef. Fariñas-Franco *et al.* (2013) stated that sediment and pseudofaeces among clumps of translocated mussels was much higher when compared to areas of laid cultch. Ragnarsson and Burgos (2012) found that the species richness in *Modiolus*/gravel/mud was nearly twofold greater compared to that of other substratum types and detailed statistical analysis tests suggested that these differences were not caused by the specific properties of the substratum itself, but rather due to the high numbers of *M. modiolus* that were found on this substrata type. Once again, the accumulation, and hence attractiveness of the habitat for its associated biodiversity is related to mature reef.

### **Box 3. Conclusions regarding the recovery potential of *Modiolus modiolus***

To date, no studies have observed recovery of *M. modiolus* beds, following either passive recolonisation after disturbance or active restoration. The process of recovery has not been documented in this species. Although regionally abundant, individual beds are often isolated or fragmented. This can effectively isolate certain populations thereby reducing dispersal and recruitment between source and sink populations. The remaining adult population (brood stock) then becomes the sole local source of gametes. In heavily depleted populations, such as in Strangford Lough (Northern Ireland), it is likely that this factor is now contributing to the limited supply of gametes. The use of a trickle spawning strategy with external fertilisation in a diminished population size will also lead to reduced fertilisation rates. Spat are gregarious during settlement. Again, depleted adult populations are also likely to lead to a decrease in optimum substrata for settlement and ultimately elevated post-settlement mortality.

Recruitment success in populations of *M. modiolus* is generally reported to be very low, even when compared to levels of bivalve recruitment generally. Juveniles of *M. modiolus* undergo rapid growth until they reach adulthood. During this period, which can take many years, mortality can be very high. Finally, the conservation value of *M. modiolus* is related to the biogenic structure generated from the mature beds. This is again dependent on recruitment and growth (adult growth is much slower than juvenile rates). Overall, several elements of the biology and ecology of *M. modiolus* significantly reduce the recovery potential of this species. These issues are exacerbated where adult populations are also diminished, highlighting the importance of brook stock size for this species. As such, restoration efforts that tackle adult decline, e.g. translocation of adults for restocking, are likely to be the most effective method of restoration.

### 3.3 Maerl beds

Maerl beds are formed by unattached coralline algae (Corallinales, Rhodophyta). Beds of maerl in the UK are often composed of two species, namely *Lithothamnion corallioides* and *Phymatolithon calcareum*. The proportion of both species varies geographically and beds may be composed of exclusively dead material. Maerl beds are a widespread marine biogenic habitat with a high conservation importance due to their associated biodiversity (Foster, 2001). Maerl forms a very complex bed structure and provides a habitat for many other species on top and within the matrix of the bed. As such, maerl beds are typically species rich with 150 macroalgal species and over 500 faunal species recorded as living on or in maerl beds (Birkett *et al.*, 1998).

Often large and extensive, maerl beds can be distributed up to the 20 m contour, although they are typically limited to shallower water by a combination of physical processes including variability in light, wave energy and seafloor slope (Bosence, 1983). Maerl beds often occur as mixed substrata and contain high proportions of gravel and dead maerl. The maerl beds themselves are often overlying softer sediments that occur just below the live maerl layer.

The morphology of maerl is highly variable depending on the depth and hydrodynamic energy experienced at the site. In faster tidal currents or areas affected by wave action, maerl thalli are often more rounded with shorter branching. By contrast, in deeper and more sheltered environments, thalli are often flatter and have a more open structure (Bosence 1976). The hedgehog form (attached to rock and branching in all directions) is also found in Loch Sween (SNH, 2014).

#### 3.3.1 Threats and sensitivity

In the context of Scottish waters, maerl is absent from the east coast of the mainland. Along the rest of the coastline, including the Shetland, Orkney and the Inner and Outer Hebrides, maerl beds are highly sensitive to several pressures that include physical changes to the sea bed (disposal of material and marine engineering), abrasion (demersal mobile/active fishing gear including trawls, dredges and hydraulic methods), the introduction of non-indigenous species (e.g. *Crepidula fornicata* which has been recorded in Scottish waters (Haskoning UK Ltd. 2006; Tyler-Walters, 2011)), enrichment (aquaculture and sea disposal), localised temperature changes and long-term, climate change (Scottish Government, 2013). Impacts from fishing and activities such as marine engineering, include both direct physical disturbance and indirect perturbations such as increased sedimentation. The physical impacts are typically associated with 1) the breakage of thalli, 2) pushing live thalli into underlying soft sediment, and 3) reduction of the large and complex interstitial spaces within the maerl matrix, which is the main contributing factor for the loss of biodiversity with impacted beds (Grall and Hall-Spencer 2003). The physical damage of thalli reduces the size and shape of maerl thalli. These smaller and simpler thalli have been found to support fewer epifaunal and infaunal species (Steller *et al.*, 2003).

#### 3.3.2 Overview of relevant studies

There are no documented studies of maerl recovery although aspects of the biology, sensitivity and associated biodiversity are well reported. Hall-Spencer and Moore (2000b) used experimental scallop dredging in an area of maerl previously unfished. Five months after the fishing event, there were 70–80% fewer live thalli when compared with samples collected prior to fishing. There were no signs of recovery in numbers of thalli or spatial coverage of maerl over the following four years. This lack of discernible recovery of a previously unfished maerl bed in Scotland over the four year monitoring period is related to the slow growth and poor recruitment of maerl species. *Phymatolithon calcareum*, which was the main species at the study site, rarely produces reproductive spores, and newly settled thalli have never been found in the British Isles (Irvine and Chamberlain, 1994).

Bosence and Wilson (2003) reviewed the maerl accumulation rates data extracted from geological samples from several studies. The value of maerl beds as habitat is a product both of the superficial living material and the interstitial spaces within the accumulation of dead maerl. The usable space within this matrix is determined by the amount of sediment infill that removes habitable interstitial spaces. Although a geological analysis, the values presented by Bosence and Wilson (2003) provide a useful indication for the rate of recovery of the whole habitat and not just the growth rate of living material at the top of the beds.

### 3.3.3 Factors influencing recovery

Six factors limit the recoverability of maerl: 1) small adult population size and hence reduced brood stock size, 2) little or no sexual reproductive output, 3) poor dispersal due to the lack of spore production, 4) reduced settlement habitat due to adult bed loss, 5) slow growth, and 6) loss of structure supporting associated biodiversity.

#### 3.3.3.1 Adult population size

The sensitivity of this habitat to human influences, such as the siltation associated with fishing activity, has significantly reduced the quality of the remaining areas. At each maerl bed, it is likely that 1) the footprint of the maerl bed has contracted, 2) the density of the thalli within a bed has declined, and 3) the average size of thalli in areas with anthropogenic disturbance has decreased. This in turn reduces the total size of the reproductive population (both for sexual and asexual reproduction) and removes dead maerl, which is in itself an important habitat. Less widely distributed and diminished beds of maerl are more likely to have reduced flexibility and health in response to environmental change.

Foster (2001) states that the enormous quantities of living fragments in many rhodolith beds suggests that fragmentation of large maerl thalli is a significant source of new recruits in established beds. As such, the density and growth rate of the existing live thalli is critically important in supporting this process.

#### 3.3.3.2 Little or no sexual reproductive output

Despite the proportion of developed, live adult thalli at some sites, the prevalence of thalli bearing reproductive tissue for spore production appears to be very low (Jackson, 2007). In the British Isles there are a few records of fertile plants. When noted, it is apparent that thalli can develop reproductive bodies at any point in the year (Birkett *et al.*, 1998). Furthermore, there are no records of new crustose forms or newly settled thalli that reproduction would produce. Even within populations where spore production within thalli is evident, Cabioch (1970) found that peaks in the development of sexual structures only occurred every 4 - 6 years for the Baie de Morlaix, Northern France. Reproduction within both species is thought to be primarily vegetative through fragmentation of the adult thalli. As recruitment is the most important component of recoverability, this apparent lack of significant spore production will limit maerl recruitment to the processes of fragmentation and subsequent regrowth. Furthermore, sexual reproduction is also associated with a dispersal advantage and an increase in genetic diversity, which is the basis of adaptation and evolution. There are certainly some advantages of asexual reproduction, i.e. only one parent is required, it typically requires less energy, no metabolic contribution is required to maintain reproductive tissues and it is often quicker (both in development time and time it takes for an individual to be contributing the population) (Lei, 2010). However, it is likely that in a recovering population, the diminished importance of sexual reproduction may have implications for the rate of population growth when compared to other species with the ability to produce large number of propagules from sexual means. It is apparent that there is little documented evidence for the sexual strategy employed by species of maerl (Jackson, 2007). The contribution of sexual reproduction to the recovery of this species is difficult to establish but presumed to be minor. Furthermore, the reliance on vegetative reproduction may have

implications for the genetic diversity within beds and hence the ability to adapt to long-term changes environmental conditions.

#### 3.3.3.3 Poor dispersal ability

Dispersal refers to the spread of some individuals away from others and may involve active or passive movements. For sessile species, dispersal is typically associated with gamete, spore or larval releases. The advantages of dispersal are considered to be 1) decrease of intra-specific competition in the adult population, 2) ability to colonize/recolonize new habitat and 3) decrease the likelihood that localized disturbance and extinction events will impact the whole population (Pechenik, 1999). Most algae produce small propagules which typically contain insufficient reserves for a prolonged planktonic existence. Despite this, the longevity of some macroalgal propagules is sufficient for considerable bouts of dispersal (Norton, 1992).

Another consequence of little or no spore production within species of maerl is a diminished dispersal capability. Reproduction relying on fragmentation of adults is clearly an effective method of reproducing locally. However, without significant spore production and associated dispersal, metapopulation processes linking source and sink populations can either be very weak or not exist. This will certainly limit the potential of these species to recolonize new areas or provide a propagule input to areas of reduced reproductive success. For many marine species in open environments, the supply of propagules from healthy source populations can be important in sustaining natural sink populations (naturally poor populations) and anthropogenically damaged source populations. Hence recoverability at a larger scale (regional and national) is often dependent on dispersal between sub-populations (Caley *et al.*, 1996).

#### 3.3.3.4 Slow growth

Bosence and Wilson (2003) reviewed the maerl accumulation rates data extracted from geological samples from several studies. The value of maerl beds as habitat is a product both of the superficial living material and the interstitial spaces within the accumulation of dead maerl. The usable space within this matrix is determined by the amount of sediment infill that removes habitable interstitial spaces. Although a geological analysis, the values presented by Bosence and Wilson (2003) provide a useful indication for the rate of recovery of the whole habitat and not just the growth rate of living material at the top of the beds. Rates of accumulation were estimated to be 0.08 m kyr<sup>-1</sup> (m 1000 yr<sup>-1</sup>) for maerl in shallow bays surrounding Orkney (Farrow *et al.*, 1984) and 0.5 m kyr<sup>-1</sup> for maerl in Falmouth (Bosence, 1976). Although not specifically measuring recovery, these studies provide an important insight into the rate of accumulation and the development of the dead matrix that contain the species rich interstitial spaces.

The growth rates for the thalli of both *Phymatolithon calcareum* and *Lithothamnion corallioides* in Spanish waters were measured at 0.486 mm yr<sup>-1</sup> and 0.015 mm yr<sup>-1</sup> respectively. Rates for thalli elongation for western Ireland populations range from 0.1 to 1.0 mm yr<sup>-1</sup> (Bosence, 1983; Foster, 2001). Growth studies and geological accretion studies indicate that maerl is extremely slow growing. Foster (2001) reviewed much of the published information on growth rates and found geographical, seasonal and environmental variation, e.g. that maerl growth below 20 m is generally an order of magnitude lower than that from shallow water. This would suggest that light and temperature are important variables for growth. Should anthropogenic disturbance disrupt any of these factors, growth may also be impacted. Successful fragmentation-based reproduction is likely to be dependent on the rates of growth and physical disturbance. Consequently either of these factors change, there could be implications for growth and fragmentation-based recruitment.

### 3.3.3.5 Loss of structure that supports associated biodiversity

Maerl beds have a rich biodiversity that includes some endemic species. This in combination with the biogenic habitat forms the whole conservation feature of interest. Surveys at Scottish sites have found 460 species of associated animals (particularly polychaetes, crustaceans, and molluscs) and 63 species of algae (BIOMAERL Team, 1999; Barbera *et al.*, 2003). A variety of factors have been identified that may influence the diversity, distribution and abundance of organisms associated with maerl beds including; 1) seasonal influences, 2) substratum, 3) ratio of living versus non-living maerl thalli, and 4) complexity of rhodolith thalli branching (reviewed in Steller *et al.*, 2003). The physical impacts of fishing and whole-scale extraction have been documented to modify the substratum, change the relative proportion of live and dead maerl and reduce thalli branching through increased fragmentation. As such, recovery of the maerl bed as a whole, including its associated biodiversity, requires maerl recruitment, development of maerl thalli structure and long-term recovery of the interstitial matrix of the bed.

Bordehore *et al.* (2003) found that fishing activity (otter trawling) increased resuspension of soft sediment from within the maerl beds and subsequent redistribution onto the upper layers of the live maerl. In turn, many ecological attributes of the associated fauna and flora i.e. number of species, abundance and biomass, are poorer in an area where physical characteristics of maerl are modified as a result of intense trawling. This elevated siltation increased the quantity of silt within the maerl bed and reduced the depth of oxygenated space (sediment and interstitial space) below the live material. Viewing maerl beds as habitats containing significant amounts of biodiversity, it is clear that whole-habitat recoverability is also reliant on the restoration of oxygenated interstitial spaces within the bed.

#### **Box 4. Conclusions regarding the recovery potential of maerl**

Based on the sensitivity of the maerl to disturbance and the six factors limiting recoverability discussed above, the potential for maerl recovery is extremely low. 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, i.e. no source and sink population connections. 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, i.e. measured in the hundreds of years. However, it should be noted that the communities associated with maerl 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.

### **3.4 Maerl or coarse shell with burrowing sea cucumbers (*Neopentadactyla mixta*)**

Maerl or coarse shell gravel with burrowing sea cucumbers (SS.SCS.CCS.Nmix, *Neopentadactyla mixta* in circalittoral shell gravel or coarse sand) has only been recorded in British waters with the majority of records being from the west of Scotland (Carruthers *et al.*, 2011). *Neopentadactyla mixta* is the most common sea cucumber within these maerl beds and typifies the 'maerl or coarse shell gravel with burrowing sea cucumber' community. This sea cucumber lives within the top layer of maerl or gravel so that only a bunch of profusely

branching tentacles around the mouth are visible when feeding. *Neopentadactyla mixta* is more widely distributed than represented by just the maerl and burrowing cucumber assemblage records, with records of the species ranging from northern Norway to the Bay of Biscay.

### 3.4.1 Threats and sensitivity

According to FEAST (Scottish Government, 2013), *Neopentadactyla mixta* in maerl or coarse shell is highly sensitive to physical disturbance and the associated habitat damage. In Scotland, this predominantly includes disturbance associated with fishing in the forms of hydraulic dredging, scallop dredging and otter trawling. Sensitivity to organic enrichment and siltation associated with aquaculture (fin and shellfish) is moderate. Sensitivity to the pressures associated with seaweed harvesting, beam trawling, line fishing, mussel and oyster dredging, military activities and tourism and recreation is considered low due to the nature of these activities (and associated pressures) and/or lack of overlap with burrowing sea cucumbers associated with maerl or coarse shell.

### 3.4.2 Overview of relevant studies

There is no documented evidence of the recovery of this community and there are no studies on its recoverability. Furthermore, there is little information on the basic biology of the dominant sea cucumber *Neopentadactyla mixta*. Jackson (2008) reviewed the biology and sensitivity of *N. mixta* and found that there was no available information on longevity, reproduction or recruitment of this or similar species. For Holothuroidea in general, spawning in temperate species is restricted to spring and summer (Pawson, 1982) and breeding is presumed to occur between April and September when the population is at the substratum surface (Jackson, 2008). Most holothurians are gonochoristic and are broadcast spawners (although some species brood their larvae). The larvae of some species show planktotrophy, others lecithotrophy, some direct development, others indirect. It is highly likely that the rarity of shell gravel and maerl substrata is an important bottleneck for recruitment of associated sea cucumber species and hence overall recoverability.

### 3.4.3 Factors influencing recovery

Three factors will almost certainly limit the recoverability of maerl or coarse shell gravel with burrowing sea cucumbers although there may be other factors; 1) all of the factors that underpin maerl recoverability due to the partial reliance on maerl gravel as a habitat constituent (Section 3.4), 2) physical habitat degradation through activities such as gravel/aggregate extraction or fishing, and 3) processes controlling recruitment of *Neopentadactyla mixta*, into coarse substratum habitats. Important recruitment factors would include adult population size, recruitment success, growth rate and reproductive strategies. As maerl provides one of the key substrata for the burrowing cucumbers, the factors influencing maerl recovery will influence recoverability of this feature as a whole. There is currently no information available for the assessment of these factors.

#### **Box 5. Conclusions regarding the recovery potential of maerl or coarse shell with burrowing sea cucumbers (*Neopentadactyla mixta*)**

Due to the lack of documented evidence for the recovery of *Neopentadactyla mixta* in maerl or coarse shell, together with a general lack of information on the biology of the species, it is not possible to draw any conclusions about the recovery potential. Whilst recovery may be partially dependent upon the distribution of maerl and coarse shell gravel, this species also occurs in other coarse, mobile sediments so whilst the recovery potential of this specific feature is unknown, there is potential for *N. mixta* to exist elsewhere, assuming that it is not subject to pressures to which it is sensitive.

### 3.5 Ocean quahogs (*Arctica islandica*)

*Arctica islandica* is the longest lived non-colonial animal known to science, with a maximum recorded (to date) life span of 410 years (Ridgway *et al.*, 2012). *Arctica islandica* is a cold water species, largely confined to temperate and sub-boreal waters with an optimum range of 6-16°C (Merrill, 1969 (in Ridgway *et al.* 2012); Cargnelli *et al.*, 1999; Witbaard and Bergman, 2003). *A. islandica* is a slow growing, long-lived specialised deposit feeding bivalve inhabiting the top 5 cm of sandy or muddy substrata, and in some cases gravel, with an optimum depth of 25-90 m (Lutz *et al.*, 1982; Witbaard and Bergman, 2003; Morton, 2011; Ridgway *et al.*, 2012), although it has been found to depths of 500 m (Morton, 2011). Suspension feeding has also been reported in this species (Cargnelli *et al.*, 1999).

#### 3.5.1 Threats and sensitivity

Due to its large size and given that *A. islandica* inhabits the top 5 cm (and deeper) of the sediment (Ridgway *et al.*, 2012), it is particularly vulnerable to disturbance associated with mobile demersal fishing gear (Rumohr and Krost, 1991). Using the bivalve *Callista chione*, Vasconcelos *et al.* (2011) found that compression and compaction of the sediments during dredging were generally associated with greater shell damage than the direct, mechanical, effects of the dredge. The mean shell strength of *C. chione* ranged from 580-3265 N (in Kg m s<sup>-1</sup>) (depending on the type of pressure applied) compared to 300-800 N for *A. islandica*, indicating that *A. islandica* is at least as, if not more sensitive to the effects of dredging (Vasconcelos *et al.*, 2011).

Whilst it is not commercially fished in the UK, it is present in areas where hydraulic dredging and scallop dredging take place and is therefore at risk of direct removal, mortality or damage from these activities. Damage, whilst not necessarily directly fatal, increases susceptibility to predation and scavenging (Hall-Spencer and Moore, 2000a; Gilkinson *et al.* 2005), parasitism and disease and may inhibit burrowing, feeding and reproductive activity. However, Ridgway *et al.* (2012) suggested that seasonal, synchronised burrowing to greater depths in the late summer and autumn may occur and that this may reduce the direct interaction of *A. islandica* with fishing gear (but not necessarily to the effects of sediment compression and compaction). These authors also suggested that the evidence for synchronised burrowing was inconclusive and this phenomenon required further study. Suggested cues for seasonal deeper burrowing included environmental factors such as temperature or anoxic bottom waters.

*A. islandica* is highly sensitive to temperature with an optimum range of 6-16°C (Merrill, 1969 (in Ridgway *et al.*, 2012), Cargnelli *et al.*, 1999; Witbaard and Bergman, 2003). Hiebenthal *et al.* (2013) reported reduced growth in *A. islandica* with an increase in temperature from 10°C to 16°C (nearing the upper thermal tolerance limit of the species (Merrill *et al.*, 1969, in Ridgway *et al.*, 2012)).

In the context of Scottish waters (based on FEAST, Scottish Government, 2013) *A. islandica* is considered to be highly sensitive to physical disturbance in the form of substratum loss, extraction, damage/abrasion or significant alteration. In Scottish waters, these pressures are largely associated with various fishing activities (and the use of mobile demersal fishing gear), offshore energy development and, to a much lesser extent, dredging and disposal activities, infrastructure development and oil and gas exploration. Hydraulic dredging and scallop dredging are highly relevant to this species with some populations also being exposed to otter trawling. *A. islandica* is sensitive to the physical pressures associated with beam trawling and aquaculture (finfish and shellfish), due to abrasion, changes in siltation and organic enrichment but with respect to aquaculture, sensitivity is considered to be low in the regions where this activity occurs. Physical disturbance (abrasion, change in habitat type, water flow rate changes, changes in siltation) is associated with offshore energy

development (particularly windfarm construction) and pipeline/cable installation in some regions of Scotland.

### 3.5.2 Overview of relevant studies

Few studies have assessed the recovery potential of *Arctica islandica* and those that have largely focus on the benthic community as a whole rather than on *A. islandica* as an individual species. Gilkinson *et al.* (2005) examined macrofaunal community development following hydraulic dredging on Banquereau (eastern Canada) and noted that, whilst recovery (of the whole community) was still ongoing at the end of the 2 year study period, bivalve recovery was considerably slower than opportunistic polychaete recovery. Furthermore, for the four commercially important bivalve species (including *A. islandica*), recruitment was low and there were no signs of recovery at the end of the study period. In this region, a 10 year fallow period is applied to dredged areas in order to enable growth of one of the key target bivalve species (*Mactromeris polynyma*) to commercial size. However, Gilkinson *et al.* (2005) emphasise that the temporal and spatial variability in recovery rates of bivalve populations and inter-specific variability must also be considered.

In a study of macrobenthic community structure in trawled and non-trawled areas of the southern North Sea, Duineveld *et al.* (2007) found that despite a fishery closure period of over 10 years, no juvenile *A. islandica* were found in the exclusion zone with populations of both species being dominated by old animals. In contrast, the number of adults/older individuals was greater in the exclusion zone than in the surrounding, fished, area. According to MarLIN (2006), the dispersal range of adult *A. islandica* is 100-1000m and it is possible that colonisation occurred from the surrounding area, particularly as this species was present (albeit at low abundances) in the fished area. Relict individuals that survived the fishing activity may also have contributed to the abundance. Lack of recruitment was attributed to low population density in surrounding areas, where populations were also dominated by older individuals (Witbaard and Bergman, 2003). Witbaard and Bergman (2003) did, however, report high densities of spat, juveniles and adults in the northern North Sea.

Ridgway *et al.* (2012) related recovery to recruitment, growth and population structure in the context of commercial fisheries. Whilst they did not directly address recovery following disturbance, they did highlight key aspects of the biological processes upon which recovery is dependent. This study emphasised that understanding recruitment was key to understanding growth and productivity of bivalves.

### 3.5.3 Factors influencing recovery

According to Cargnelli *et al.* (1999) the planktonic development (eggs and larvae) stage appears to be temperature-dependent, lasting between 32 and 55 days (at temperatures of 13 and 8-10 °C, respectively) before settlement. Growth of juveniles is rapid but overall, growth is slow after this initial phase (Morton, 2011) and is dependent on temperature, primary production, depth, substratum characteristics (Witbaard *et al.*, 1999), river discharge (depending on proximity to the coast and inputs of freshwater) and the North Atlantic Oscillation (Carroll *et al.*, 2009). Age at sexual maturity is variable but in a review of the biology of *A. islandica*, Cargnelli *et al.* (1999) reported a minimum of 7 years and an average of 12.5 and 13 years for males and females, respectively. This strategy of initial fast growth and late maturation enables *A. islandica* to quickly increase its size beyond that accessible to predators thus ensuring longevity (Ridgway *et al.*, 2012). Ridgway *et al.* (2012) indicated that both temperature and density-dependent factors have been proposed as explanatory processes in recruitment and longevity of *A. islandica*, with a prolonged reproductive life span being associated with limited opportunities for recruitment.

Spawning is protracted (spring to autumn) although recruitment of *A. islandica* is low (Cargnelli *et al.*, 1999) and unpredictable or sporadic. Harding *et al.* (2008) found recruitment to be related to temperature, with poor recruitment being associated with months where above average temperatures between January and March were recorded and, that recruitment periods alternated with several years of no recruitment. Butler *et al.* (2009, in Ridgway *et al.*, 2012) reported long periods of no recruitment in the waters around the Isle of Man. Powell and Mann (2005) reported spatial variability in *A. islandica* recruitment in the mid Atlantic Bight (Hudson Canyon, Long Island, USA) with one area showing a near absence of recruits, one area showing a prolonged absence of recruitment and another area showing widespread presence of recruits. Their study supported the theory of low recruitment success in long-lived species with successful recruitment occurring only at carrying capacity. Witbaard and Bergman (2003) reported alternation between periods of highly successful recruitment and years of no recruitment but were unable to explain this variation. They proposed various hypotheses including low abundance of reproductively active adults, thermal stratification in the water column, trawling impacts, high mortality of settled spat and changes in sediment structure, but these have yet to be tested. A further source of variability is the selectivity of sampling gear.

Ridgway *et al.* (2012) documented almost continuous recruitment (based on annual growth rings) of *A. islandica* in Belfast Lough but reported that small clams were rare in their samples (from Belfast Lough) and that this was comparable to other populations throughout the eastern American and Icelandic coasts. It is not clear whether this is an artifact of the sampling technique or whether it represents recruitment failure during the sampling period.

#### **Box 6. Conclusions regarding the recovery potential of *Arctica islandica***

There are no documented studies showing clear recovery of *A. islandica* following an initial decline. Studies documenting substantial recruitment of this species are mostly lacking, although Ridgway *et al.* (2012) did report continuous recruitment for Belfast Lough, Northern Ireland. The combination of high sensitivity, slow adult growth, low population density, sporadic spawning and resulting poor recruitment indicate that natural recovery of this species would occur over very long time-scales, possibly hundreds of years. This estimate is based on the desire that a recovered population would ideally contain the full breadth of potential age classes, i.e. contain individuals approaching the maximum reported ages for this species (>400 years). Concentration of the adult brood stock and hatchery techniques are the only potential appropriate methods for active restoration.

### **3.6 Soft sediment habitats and communities**

Very little information on the recovery of species (of relevance to this study) inhabiting soft sediment habitats was found. Furthermore, recovery is often dependent upon physical recovery the habitat and there are a number of common processes and factors that apply to multiple soft sediment habitats. Therefore, the two soft sediment features of interest to this review (burrowed mud and shallow tide-swept coarse sands with burrowing bivalves) have been considered together in this section.

#### **3.6.1 Burrowed mud**

The 'burrowed mud' protected feature incorporates the biotopes:

1. Seapens and burrowing megafauna in circalittoral fine mud (SS.SMu.CFiMu.SpNMeg), with seapens (*Virgularia mirabilis*; *Pennatula phosphorea*), burrowing crustaceans (*Nephrops norvegicus*) and the anemone *Cerianthus lloydii*, together with epifaunal decapods and Ophiuroidea and significant densities of infaunal bivalves, polychaetes and

echinoderms (*Brissopsis lyrifera*). Evidence of heavy bioturbation in the form of burrows and mounds are a characteristic feature of this biotope (Connor *et al.*, 2004).

2. Burrowing megafauna and *Maxmuelleria lankesteri* in circalittoral mud (SS.SMu.CFiMu.MegMax) characterised by the decapod crustaceans *N. norvegicus* and *Callinassa subterranean*, the echiurian *M. lankesteri* and low numbers of *V. mirabilis* (Connor *et al.*, 2004).

Species of particular interest which are also part of burrowed mud include the scarce fireworks anemone *Pachycerianthus multiplicatus* and the tall seapen *Funiculina quadrangularis*.

The communities characterising burrowed mud habitats are typically found in stable, fine sediments (sand and mud) in fully marine conditions (Greathead *et al.*, 2011). In Scottish waters, seapens (*V. mirabilis*, *F. quadrangularis* and *P. phosphorea*) generally occur in the deep basins of sea lochs, the North and South Minches, the Outer Hebrides, the Firths on the east coast of Scotland and the Fladen area (offshore to the north-east of Aberdeen) of the North Sea (Greathead *et al.*, 2011).

#### 3.6.1.1 Threats and sensitivity- Burrowed mud

Burrowed mud habitats and their associated fauna, are suggested to have a high sensitivity to large scale physical change of the sea bed, as a permanent change to another seabed habitat type other than fine sediment is likely to lose the characterising community. The mud substrata habitats are also predicted to have a medium sensitivity to organic enrichment, removal of target and non-target species, siltation changes, sub-surface abrasion/penetration, surface abrasion and variation in the current and wave energy regimes locally (Scottish Government, 2013).

#### 3.6.2 Shallow tide-swept coarse sands with burrowing bivalves

This habitat includes *Moerella* spp. with venerid bivalves in infralittoral gravelly sand (SS.SCS.ICS.MoeVen) and is typically found in tide-swept areas on open coasts and in the mouths of sea lochs, subject to moderately strong tidal water movement. The substratum is composed of coarse and gravelly sands. This habitat is found in the infralittoral zone (shallow subtidal). As it is also characterised by the presence of algae, much of this habitat is within the photic zone. Typical species include the bivalves *Moerella pygmaea* or *M. donacina*, *Dosinia lupinus*, *Timoclea ovata*, *Goodallia triangularis*, *Chamelea gallina* and *Spisula solida*, together with amphipod crustaceans and nephtyd and spionid polychaetes (Connor *et al.*, 2004).

Suspension feeding bivalves dominate and are abundant in the coarse sediment. Infaunal species also include tanaids (shrimp-like crustaceans) and sand hoppers (Talitridae). The bivalves may be preyed upon by starfish, snails and flatfish. Seaweeds including sugar kelp (*Saccharina latissima*) and *Ulva* spp. may be present on the sediment surface attached to small pebbles and larger shells.

#### 3.6.2.1 Threats and sensitivity - Shallow tide-swept coarse sands with burrowing bivalves

Shallow tide-swept coarse sands with burrowing bivalves are suggested to have a high sensitivity to gross change or loss of the physical habitat (associated with the total loss of habitat). The bivalves associated with this habitat type are suggested to have moderate sensitivity to substratum removal, removal of target species (associated with targeted *Spisula* and *Ensis* removal by mechanical or hydraulic dredges) and sub-surface abrasion/penetration, related to dredging and fishing methods employing mobile demersal gear. Shallow tide-swept coarse sands with burrowing bivalves are proposed to have a low

sensitivity to organic enrichment, removal of non-target species, localised salinity changes, surface abrasion, temperature changes, hydrodynamic modification and wave exposure adjustments locally (FEAST, Scottish Government, 2013).

### 3.6.3 Overview of relevant studies

Both 'burrowed mud' and 'shallow tide-swept coarse sands with burrowing bivalves' are combinations of a soft substrata habitat with distinctive benthic communities. Recovery is dependent on the physical presence and structure of the habitat and its colonisation by burrowing fauna. Little information is available for the recovery of most of the species present in these habitats. Edwards and Moore (2008) examined the reproduction of the seapen *Pennatula phosphorea* collected from sea lochs in the west of Scotland. This species was observed to use a broadcast spawner reproductive strategy and external fertilisation of gametes. The average oocyte diameter during the period of greatest spawning (June to August) was large and suggests that larvae will be lecithotrophic whilst in suspension. These authors indicate that the large egg size may suggest prolonged residence in the planktonic phase, resulting in greater dispersal potential when compared to other species. If this were the case, it implies that connectivity between sub-populations is high (depending on the hydrodynamic conditions and distance between sub-populations), facilitating recovery at a regional level. However, *P. phosphorea* is present in sea lochs where residence time may be high, reducing the dispersal potential of pelagic larvae. Therefore, larvae may be retained within the vicinity of the spawning population and connectivity between populations may be compromised. Fecundity of this species, based on the expulsion of larger oocytes during the spawning period, is estimated to be modest and between 3000 and 8000 oocytes per colony.

The reproductive behaviour of *Funiculina quadrangularis* is similar to that of *P. phosphorea* except the main spawning period is late autumn and winter (Edwards and Moore, 2009). This is suggested to be beneficial as settlement and metamorphosis of this species can occur well before the spring bloom. The average oocyte diameter during the spawning period was large (>800 µm). The larger diameter, and hence greater stored nutrients, again indicates a lecithotrophic existence in the plankton and significant residence time (Edwards and Moore, 2009). Such strategies are likely to reduce the value of resident adults contributing to the recovery of the source population and increase the strength of meta-population dynamics in providing recruitment to colonised sites. Residence time near the source population is likely to be increased within sea lochs.

Limited information was found regarding the recovery potential of *Maxmuelleria lankesteri* although, based on a survey in Loch Sween, Hughes *et al.* (1996) suggested that, despite apparently high fecundity (90,000 eggs recorded from the gonoduct of a single individual), recruitment in this species may be sparse and infrequent. It was suggested that reproduction may occur by parthenogenesis (embryo development from an unfertilised egg) (Hughes *et al.*, 1996) but given the lack of evidence of recruitment, the recovery potential of this species is likely to be low. This would be further reduced in populations in sheltered sea lochs where the residence time of the water may be high.

No studies indicating the recovery potential of 'shallow tide-swept coarse sands with burrowing bivalves', as a community or habitat, were found although component species of this biotope, such as *Spisula solida*, are commercially important and have, in some parts of the world, been over-fished. There is therefore a limited amount of literature concerning restoration efforts to enable the recovery of commercially viable stocks. Joaquim *et al.* (2008) indicated that, in combination with management measures to prevent losses, transplantation of *Spisula solida*, as a bivalve stock enhancement strategy, could potentially enable recovery of an economically viable fishery. Survival one year after transplantation was around 50% and this was broadly consistent with survival rates documented by other

authors (Joaquim *et al.*, 2008 and references therein). Despite the apparently high mortality rate, it is of note that *S. solida* reaches sexual maturity within 1 year and generally only lives for around 2-3 years (up to 10 years in some cases) (Annex 2). Therefore, survival at this level is likely to be sufficient to enable spawning, settlement and possible recruitment. Whilst the re-establishment of commercial fishery is not a priority for SNH, this study indicates that, at least some components of this biotope have the potential to recover. Any transplantation effort would need to consider predation, density dependent effects, individual size and other pressures which may influence bivalve survival. It is of note that attempts to restore bivalve populations have frequently been unsuccessful because the cause of the decline has not been addressed, coupled with inappropriate habitat conditions, low population density and life history traits that automatically restrict recovery potential (Joaquim *et al.*, 2008).

Joaquim *et al.* (2008) indicated that survival following transplantation may be related to the ability of individual clams to burrow rapidly after transplantation, in combination with density dependent factors and predation. Alexander *et al.* (1993) examined the burrowing rate of 21 species of bivalve in various sediment types ranging from fine gravel to mud and concluded that species could be classed as 'substratum generalists (wide range of grain sizes)', 'substrate specialists (narrow range of grain sizes)' or 'sensitive (showing characteristics of both generalists and specialists with burrowing ability being restricted to a defined but not necessarily narrow range of sediment classes) but that the burrowing speed of all species decreased as the sediment grain size became more unfavourable. Species of relevance to the SS.SCS.ICS.MoeVen biotope (and the associated shallow *Venus* community) and others of relevance to this study (e.g. *Zostera* biotopes) include *Spisula solida* (associated with SS.SCS.ICS.MoeVen), *Cerastoderma edule* and *Abra alba*, all of which were classified as 'specialists' with burrowing ability being restricted to a narrow range of sediment classes. Sensitive species included *Ensis* sp. (relevant to SS.SCS.ICS.MoeVen), *Tellina tenuis*, *Macoma* sp. and *Scrobicularia plana* whilst generalists included *Venus* spp. *Abra alba* and *Mya arenaria*. Alexander *et al.* (1993) concluded that burrowing ability is directly related to the ability to avoid predation and maintain contact with the overlying sediment and water column, necessary for feeding, ejection of faecal matter, respiration and burrow ventilation.

The degree of substratum selectivity displayed by the various species will ultimately affect their potential for recovery following disturbance. Physical disturbance, such as that associated with mobile demersal fishing gear or construction activities (e.g. windfarm construction, cable installation) may result in changes to the overall particle size or the textural characteristics of the sediment by, for example, mixing and/or resuspension and deposition of fine particles. Indeed, Constantino *et al.* (2009) noted that sediments were coarser in areas influenced by clam dredging but acknowledged that this degree of change to the substratum could also be caused by storms. Species classed as generalists may well be able to re-establish themselves and burrow into the sediment but sensitive or specialist species may not. This habitat specificity may also apply to larval settlement. Therefore, whilst the recovery potential of some species may be high, either through natural settlement or translocation, the recovery potential of the community as a whole may not be.

#### 3.6.4 Factors influencing recovery

Due to the limited amount of information, recovery of most of the specific infaunal species associated with 'burrowed mud' and 'shallow tide-swept coarse sands with burrowing bivalves' has not been documented in previous studies. Therefore, this section has drawn upon other examples of mud and sand communities reported in the literature.

Soft sediment habitats have interconnected physical, chemical and biological components, although the importance of the chemical component diminishes as particle size increases. The physical aspects of the sediment include the distribution of particle size, sorting and kurtosis. The chemical characteristics of the sediment include the organic carbon and

nitrogen concentrations, redox or sulphide states and the concentration of pollutants (Dernie *et al.*, 2003). The biological component will include the microflora, meiofauna, macrofauna and megafauna (both resident and transient). All of these components are interdependent and must be present in a suitable condition for the overall habitat to be considered in a normal or recovered condition.

The recovery of soft sediment habitats depends on the type of initial impact and to what extent that pressure is removed during the recovery phase. For example, the impact longevity for physical disturbance in a mobile, coarse sand habitat may be short whereas mud habitats may remain contaminated with heavy metals for extremely long periods, which can significantly hinder faunal recolonisation.

Although the succession of faunal components is typically well documented in the literature, the recovery of the occupied habitat in terms of the physical and chemical components is often poorly reported. Particle size parameters are often provided with studies detailing the recovery of infaunal communities following disturbance. However, particle size statistics reported are often to facilitate the explanation of the biological data and not as a stand-alone analysis of physical habitat recovery. Dernie *et al.* (2003) also makes the point that the biological recovery rate is linked to the restoration of the sedimentological characteristics of the habitat and that this process is not well described by physical characteristics such as grain size distribution alone, hence the recovery of habitats are poorly described when compared with the biological components. The following examples of recovery are therefore dominated by studies of infaunal and epifaunal succession.

In coarse, gravelly sediments, physical recovery times (in terms of the disappearance of dredge tracks) can take up to 20 years (Foden *et al.*, 2009). However, the importance of these topographic changes to biological community development, in comparison to recovery of sediment structure in terms of particle size distribution and organic content (for example), is not clear.

Kaiser *et al.* (2006) found that predominantly sandy habitats had relatively short recovery periods following disturbance (based on number of species/taxa and abundance), suggesting that coarser substrata habitats have naturally high rates of physical disturbance. The adaptation of the community to these conditions may mean that it is better able to resist and subsequently recover from the physical disturbance associated with fishing activity. Gilkinson *et al.* (2005) examined the recovery of sandy communities following experimental dredging treatments. There were immediate declines in abundance and biomass of most taxa after the dredging, followed by significant increases in the abundance of polychaetes and amphipods for the first 12 months. For the following 12 months the abundance of opportunistic species also increased. Gilkinson *et al.* (2005) concluded that after 24 months the infaunal community was still in a colonising phase. The large bivalves associated with this particular sandy sea bed were particularly slow to recruit and showed no sign of recovery over the 2 year period.

Dernie *et al.* (2003) compared the recovery of various intertidal sandy and muddy habitats following physical disturbance and found that the rate of recovery of benthic infaunal communities was linked to bed load transport. Clean sand communities had the most rapid recovery following disturbance with recovery rate being predominantly determined by physical processes. Recovery in such habitats was rapid and often driven by relatively mobile species common to sand habitats. Dernie *et al.* (2003) found that physical recovery of mobile sand habitats is rapid and occurs within 100 days post-disturbance. Bed load transport may also be an important mechanism for recolonisation of disturbed areas. Beukema *et al.* (1999) also highlights the importance of bed load transport as a recovery mechanism for certain species. It is believed that the movement of adults is more important than larval recruitment in coarse habitats. As significant bed load transport is a feature in

high energy, coarse substrata habitats, it is likely that this process facilitates the recovery of these habitats through the supply of new bed material and adults.

The potential for recovery for sandy habitats following dredging has been found to vary depending on the amount of material removed from the site. Desprez (2000) found that aggregate extraction in Dieppe (France) resulted in a shift from a predominantly coarse sand seabed to a fine sand substrata. This also changed the successional end-point for the infaunal community making recovery to a previous state impossible. Boyd *et al.*, (2005) also state that the recovery of sandy habitats is only possible if the topography and original sediment composition are restored. By contrast, Newell *et al.*, (1998) suggests that sand and gravel habitats are often dominated by long-lived K-selected species and that the community succession is held at the transitional phase for longer than in finer sediments.

With regard to mud substrata habitats, Newell *et al.* (1998) examined benthic macrofauna following dredging operations. Recovery ranged from up to one to three years in areas of high current velocity and 5-10 years in areas of low current velocity. Dernie *et al.* (2003) also found that communities from muddier habitats had the slowest physical and biological recovery rates when compared to habitats dominated by coarse sediments. These findings were also found earlier by the meta-analysis undertaken by Collie *et al.* (2000).

Macleod *et al.* (2004) examined the recovery of a soft mud sediment habitat following the removal of fin fish aquaculture cages and the cessation of associated organic enrichment. Recovery monitoring examined physical, chemical and biological variables. The time for recovery was found to be different for each of the parameters when assessed individually. Sulphide levels at the site diminished rapidly and were indistinguishable from background levels after 24 months. Organic matter within the sediment remained elevated for the three year study period whilst other chemical indicators suggested progressive recovery during the same period. This suggests that much of the remaining organic matter was refractory and unavailable for the biota. As such, Macleod *et al.* (2004) suggests that organic enrichment is a poor indicator of soft sediment recovery. Although the overall trajectory of the community was towards recovery, the benthic data for the site indicated an impacted community even 36 months after the removal of the fish cages. This study highlights the variability in recovery assessments depending on the variables used. As soft sediment habitats are dependent on interlinked physical, chemical and biological aspects, recovery may well be determined by the parameter slowest to recover.

Dernie *et al.* (2003) presented Figure 3 to indicate the relative importance of physical, chemical and biological processes in driving the recovery of habitats of differing silt/clay fraction. As stated above, the recovery of sandy substrata habitats (low silt/clay) are often dominated by physical processes, e.g. bed load transport and storm resuspension for infill. In areas of muddy-sand and sandy-mud, the recovery process can be heavily influenced by biological factors, e.g. stabilisation of the sediment by ephemeral tube-building polychaete colonisation. Finally, in the finest sediments with the greater proportion of silt/clay, the recovery is often driven by chemical processes connected to oxidative states and organic enrichment.

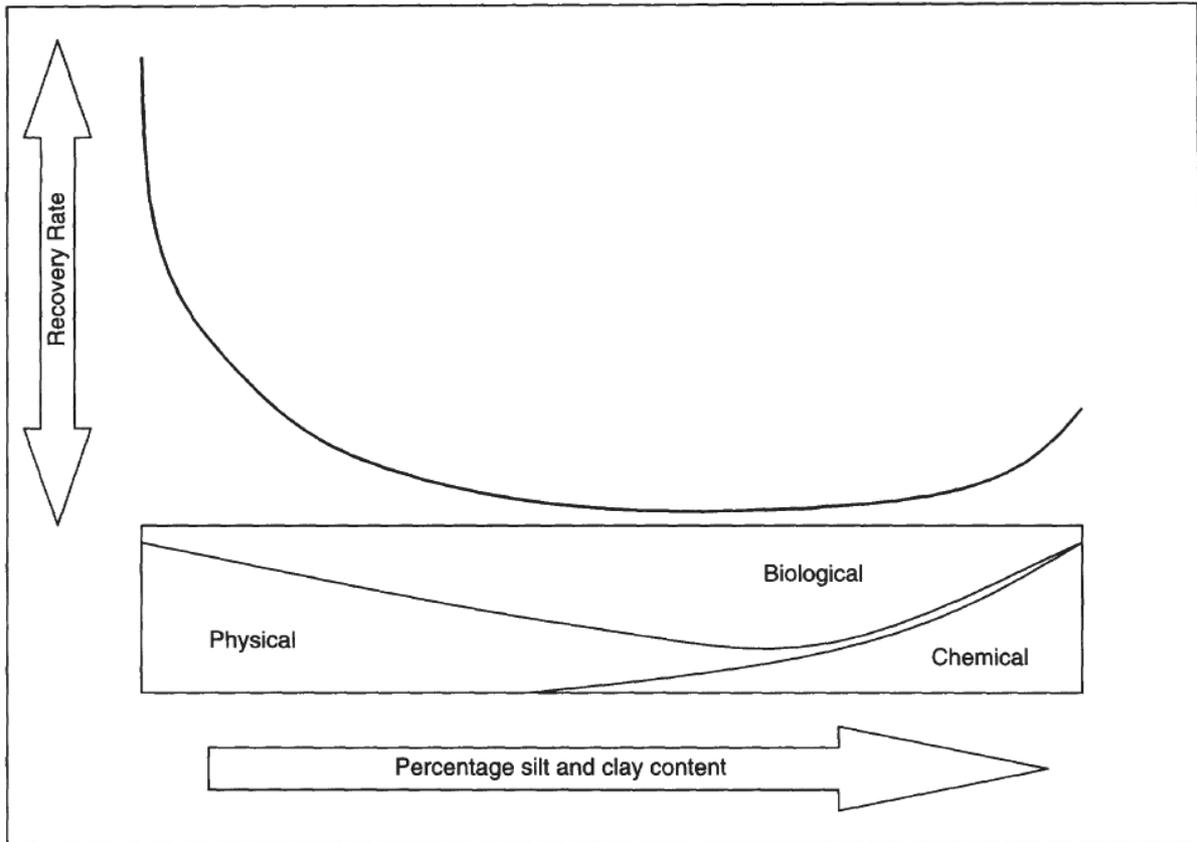


Figure 3. Schematic diagram from Dernie et al. (2003). In habitats with a low silt and clay content, physical processes dominate recovery, which may be rapid. With increasing silt and clay content, biological recovery rate is reduced as chemical and biological factors act to increase sediment stability. In silt and clay dominated sediments, the recovery rate increases slightly as the substratum acts as a fluid and aids recovery of the biota through 'slumping' of adjacent sediments and organisms into disturbed pits.

### **Box 7. Conclusions regarding the recovery potential of burrowed mud and shallow tide-swept coarse sands with burrowing bivalves**

Whilst there are no studies documenting recovery, it is likely (given evidence presented here) that recovery could occur but timescales are unclear due to the many physical, site-specific variables. There are no documented studies showing clear recovery of the sediment habitats examined. However, some species associated with burrowing bivalve communities have shown recovery potential through studies of active restoration. It is of note that recovery for many bivalve species is dependent on the sediment type and, amongst other things, the influence of grain size on burrowing ability. Therefore, given the evidence presented here, the recovery process could occur but timescales are determined by these key environmental variables.

Recovery processes and rates for similar habitats that occur on the sand to mud substrata gradient initially appear to be highly variable and will depend on many factors. The initial condition is clearly set by the type, intensity and duration of the impacting pressure. The subsequent recovery of the habitat is dependent on the environmental factors present at the site and local supply of larvae for recruitment (Thrush *et al.*, 2008). For example, mud habitat recovery has been suggested to be faster at higher temperatures, which is presumed to relate to the rate of microbial processes that remove excessive organic matter (Black, 2001 in Macleod *et al.*, 2004). Furthermore, local hydrographic conditions influence recovery rates and in sheltered areas recovery can be significantly longer than in higher energy environments (Black, 2001 in Macleod *et al.*, 2004). The timescale for recovery will also depend on whether recovery of the habitat is a required prerequisite for community development or whether community succession can occur at the same time as habitat recovery.

### **3.7 Fan mussel aggregations (*Atrina fragilis*)**

The fan shell (*Atrina fragilis*, previously *Actrina pectinata* and *Pinna fragilis*) is a large (30 – 48 cm in length) benthic triangular mussel, which inhabits mud, sand and gravel habitats in the lower intertidal and subtidal, from low water to depths of 400 m (Solandt, 2003). However, a clay-sand mix sediment at 30 to 50 m in depth is suggested as the preferred habitat for this species (Fryganiotis *et al.*, 2013). The fan mussel lives with its anterior (pointed end) semi-submerged (half to three quarters of shell) in the sediment, with its posterior (broad) end protruding from the surface (Solandt, 2003). This broad end gapes slightly, enabling the pale edge of the mantle to emerge and the mussel to filter feed through its inhalant and exhalant siphons (Clark, 2012). The mantle secretes new shell material around the upper edge as the mussel grows (Clark, 2012). The anterior end is anchored in the sediment by fine byssus threads attached to small stones and/or shell fragments (Solandt, 2003; Clark, 2012).

*Atrina fragilis* is described as 'scarce' and are often solitary, but populations can occur as small groups or patches of individuals forming small beds or aggregations (UK Biodiversity Group, 1999). However, almost every reported record in the 20<sup>th</sup> century is based on sightings of individuals (Solandt, 2003). This is in contrast to the historical record, which suggests that *A. fragilis* occurred and were caught in dense aggregations in shallow UK waters around the middle of the 19<sup>th</sup> century (Solandt, 2003). It is proposed, however, that dense aggregations of *A. fragilis* may still exist in deeper protected waters that are not disturbed. This is the case in the Small Isles MPA, where underwater video footage showed that the fan mussels occurred as scattered individuals or in small clumps, in poorly sorted mixed muddy sediments (biotopes - SS.SMu.CFiMu.MegMax and SS.SBR.SMus.Afrag) in depths greater than 100 m (102 - 274 m) (Clark, 2012). A variety of species were observed living on, or in close association with *A. fragilis*, indicating that it may be an important

'structural' species in sedimentary habitats with the emergent shells providing a hard substratum for sessile species (e.g. barnacles, tubeworms etc.) and encrusting seaweeds (Warwick *et al.*, 1997; Clark, 2012). The presence of *A. fragilis* on scallop grounds may be of particular importance to the fishery since juvenile pectinids have been recorded attached to their shells in the Adriatic (Hall-Spencer *et al.*, 1999).

It has been noted that the faunal community associated with aggregations of *A. fragilis* show some similarities to a sparse horse mussel *Modiolus modiolus* bed, (Clark, 2012) and as such the latter could provide a source of information on the expected faunal composition and the potential for recovery for the faunal community associated with *A. fragilis*.

### 3.7.1 Threats and sensitivity

Scottish Government (2013) provides an overview of the sensitivity to pressures of *A. fragilis* aggregations in Scottish waters. *A. fragilis* is most sensitive to physical change to the sea bed (i.e. a change in substratum type, physical removal of substratum and abrasion), changes in siltation rates (high) and removal of non-target species. These pressures arise from fishing (including hydraulic dredging, otter trawling and scallop dredging) and activities including infrastructure development, dredging and extraction of sand and gravel. *Atrina fragilis* is considered to have low sensitivity to low-level changes in siltation rates (as a result of substratum extraction and fishing techniques) and organic enrichment associated with aquaculture and waste disposal (industrial and agricultural, sewerage and sediment disposal) (Scottish Government, 2013). Potential impacts can occur as a result of removal of or change in habitat type, direct impacts on biological function (siltation of syphon and gills) and direct removal, lethal damage or mortality (Scottish Government, 2013).

Local changes in water flow (tidal current) and wave exposure (as a result of on-shore and off-shore infrastructure, seabed substrate and water extraction) and abrasion of the sea bed surface (through fishing activities, seaweed harvesting) could potentially be damaging to *A. fragilis* aggregations (Scottish Government, 2013). However, due to the ability of *A. fragilis* to recover from damage to its upper surface, the potential for lethal damage is lower than that associated with sub-surface abrasion (i.e. creeling and potting which only causes surface abrasion versus otter trawling and scallop dredging which results in sub-surface abrasion/penetration).

Bottom trawling has long been documented as a major disturbance for a variety of benthic organisms, including *A. fragilis*, with considerable anecdotal information confirming trawler damage as a probable cause of population decline in open sea (Hall-Spencer *et al.*, 1999; Solandt, 2003). *A. fragilis* is ranked as a highly susceptible species to physical disturbance and bottom trawling due to its fragile shell, longevity, size, and inability to re-burrow into sediment after a disturbance event (Hall-Spencer *et al.*, 1999; Solandt, 2003).

*Atrina fragilis* is adapted to limit damage to its upper surface from minor disturbance, as most soft tissues of the mantle are withdrawn into the posterior margin of the shell upon contact with the outer surface (Solandt, 2003). The shell, however, cannot be completely closed, rendering the mussel potentially susceptible to smothering (e.g. from spoil dumping) (Clark, 2012). Young and Thomson (1976) suggested that post damage to the posterior (upper) portion of the shell, regrowth occurs at the rate of 1cm per year. This may have had some impact on the ability of *A. fragilis* to re-generate shell material after passage of the older, lighter beam scallop-fishing trawlers and dredges in the past (Solandt, 2003). However, modern toothed heavy beam trawls and dredges dig into the sediment and either remove the whole organism (e.g. speared on trawl teeth) or lethally damage the soft tissue in the inner part of the shell (Hall-Spencer *et al.*, 1999; Solandt, 2003). As demonstrated by Hall-Spencer *et al.* (1999) trawling has caused over 70% reduction in the *A. fragilis*

population of the Adriatic (Gulf of Venice), where 90% of the specimens caught by trawlers were lethally damaged.

Anecdotal records from divers suggest that considerable numbers of *A. fragilis* may be found in areas where scallop trawlers and dredgers cannot set their gear, for example in waters of high current speed, where the sea bed is more steeply sloping, or where there is a narrow, yet deep stretch of water, where trawlers cannot run a trawl before having to turn (Solandt, 2003). It is of note that this seascape is indicative of the Western Isles of Scotland, particularly the geomorphology of the Sound of Canna, where the largest aggregations of *A. fragilis* have recently been recorded (Solandt, 2003; Clark, 2012).

### 3.7.2 Overview of relevant studies

The ability of *A. fragilis* aggregations to recover following disturbance is relatively unknown due to the rarity of this species and scarcity of studies investigating evidence of recovery. Fryganiotis *et al.* (2013) provide the only research into the recovery potential of *A. fragilis* following disturbance. This study compared distributions and abundances of *A. fragilis* in two contrasting areas in the north Aegean Sea (a trawled area and an area where trawling had been prohibited). Very little information is known about the biology of *A. fragilis* (and thus the factors influencing recovery) with available information mainly limited to species distribution and dynamics, with many assumptions made based on the biology of similar species (Solandt, 2003; Fryganiotis *et al.*, 2013).

### 3.7.3 Factors influencing recovery

The growth rate of *A. fragilis* is around 3-4 cm per year (UK Biodiversity Group, 1999a) with a lifespan estimated to extend beyond 12 years (Fryganiotis *et al.*, 2013). The closely related Mediterranean species, *Pinna nobilis*, is reported to live for up to 18 years (Butler *et al.*, 1993). Oceanographic differences between sites, however, have a considerable bearing on growth rates and population of Pinnid bivalves, dependent on the velocity and organic content of the water currents (Shimeta and Jumars, 1991). The onset of sexual maturity decreases the growth rate of the shell, due to energy expenditure directed at gonad development (Solandt, 2003). Therefore, age of maturity and speed of reaching reproductive size will likely have considerable impact on subsequent growth rates (Richardson *et al.*, 1999).

A feature of the biology of the fan mussel that likely limits its distribution is the stochastic nature of its recruitment success (Solandt, 2003). The fan mussel is a marine broadcast spawner, so fecundity is likely to be considerable within the species, although nothing is known about the spawning timings, egg and sperm abundance or ratios of *A. fragilis* (Solandt, 2003). Fertilisation success is likely determined by currents, timing of spawning and the proximity of other conspecifics. Recruitment success is likely determined by the availability of food, appropriate habitat for larval settlement and oceanographic effects (Cowen *et al.*, 2009). Once populations of fan mussels have been reduced by disturbance, it is very difficult for them to recover without sufficient local populations to ensure gamete fertilisation (Solandt 2003). It is speculated that shallow water recruitment relies on larval production in deep water populations (Solandt, 2003). Two individuals of *Atrina fragilis* were successfully translocated to Loch Carron in north-west Scotland suggesting that translocation may be an option for recovery of sites by providing local conspecifics in an area of high current for larval dispersal (Solandt, 2003). However, the two individuals were believed to have been subsequently removed by inquisitive divers (J-L Solandt, Marine Conservation Society, pers. comm. Sept 2014). As such there is evidence that translocated individuals can survive (though limited to this one study) but the period of survival is not known. Furthermore, there is no information on reproductive output and thus estimates of recovery potential. The translocation of a single individual of *A. fragilis* within Northern

Ireland was initially considered successful based on survival of the translocated individual. However, subsequent attempts to find the translocated individual failed and it was presumed to have died (J. Breen, Environment Agency Northern Ireland, pers. comm. April 2013).

Fryganiotis *et al.* (2013) compared distributions and abundances of *A. fragilis* in two contrasting areas of the north Aegean Sea, one which is intensively trawled and one where trawling has been prohibited for over 25 years. They recorded decreased size and density of *A. fragilis* in the trawled, compared to non-trawled areas, suggesting a temporal aspect to the recoverability of this species. However, in the non-trawled area, population structure within the *A. fragilis* aggregations was poor with a complete absence of small, juvenile specimens indicating poor recruitment (all specimens caught were aged between 7 to 16 yrs) (Fryganiotis *et al.*, 2013). Whilst not addressed by the authors, this suggests that even in protected areas where disturbance is prohibited and aggregations of mature specimens occur, that recruitment (and thus population structure) in the local area may be sporadic and/or influenced by additional currently unknown factors that occurred or were modified within the preceding 7 year period (e.g. oceanographic effects including the velocity and organic content of water currents, eutrophication, the location of local conspecifics etc.).

#### **Box 8. Conclusions regarding the recovery potential of *Atrina fragilis***

There is evidence to suggest that recovery of *A. fragilis* may be possible following removal of the physical disturbance. However, recovery may be lengthy, sporadic or unachievable depending upon local oceanographic conditions (i.e. velocity and organic content of water currents), the proximity of other conspecifics and sediment structure. Recovery is more likely where disturbance is limited and/or patchy in extent, there are viable local populations of conspecifics (for gamete fertilisation and larval availability), the sediment structure is maintained (to enable larval settlement and growth) and oceanographic conditions are favourable (i.e. water current direction, velocity and organic content). There is limited evidence that translocated individuals can survive, for an unknown period of time, but the long term survival and potential for recovery of this species is not known.

### **3.8 Native oyster (*Ostrea edulis*)**

*Ostrea edulis* is a species of oyster native to Europe and commonly known as the European flat oyster. Individuals of *Ostrea edulis* can be as old as 10-15 years, although the majority of individuals are between 2-6 years old (Jackson and Wilding, 2009). Sexual maturity occurs in the third year of growth as protandrous alternating hermaphrodites and regularly switch between egg and sperm production. Gamete maturation in the UK begins in March or April and is in part temperature dependent, with the period of gametogenesis being greater in warmer conditions. Fecundity may be as high as 2,000,000 in large individuals (Jackson and Wilding, 2009).

*Ostrea edulis* can occur in both intertidal and subtidal habitats, and will colonise all substrata from mud to coarse ground. Spat will also settle and grow on man-made structures and will settle on many hard surfaces, e.g. compacted silt, muddy gravel with shells, sand and rocks (Laing *et al.*, 2006) although the species have shown a clear preference for shell material over gravel or pebbles where survival is higher (UMBS, 2007 in Shelmerdine and Leslie, 2009). Most oyster species show gregarious settlement behaviour and preference for conspecific shell material. The growth rim of the adult *O. edulis* has been reported by Korringa (1941) to provide the best settling surface for recruiting spat of this species. *Ostrea edulis* is also tolerant of salinities reduced to as low as 18 PSU (Jackson and Wilding, 2009).

### 3.8.1 Threats and sensitivity

In the context of Scottish waters, *O. edulis* is considered to be highly sensitive to physical change, loss or removal of the substratum, removal of target and non-target species, changes in siltation rate, the introduction of pathogens, the introduction of non-indigenous species, organic enrichment, and changes in wave exposure. The species is moderately sensitive to local water flow changes, temperature change, surface abrasion and genetic modification (Scottish Government, 2013).

As a species with a long history of commercial exploitation, there has been a dramatic reduction in the population of *O. edulis* in the UK since the middle of the last century (Laing *et al.*, 2006).

There are four principal threats to populations of *O. edulis*. The most important threat is that of over-exploitation. Within Scottish waters, there is a long tradition of gathering *O. edulis*, particularly during the 19<sup>th</sup> century when landings were at their greatest. The review by University Marine Biological Station Millport (2007) provides a thorough overview of the boom and bust fishery for *O. edulis* in Scotland. Apart from a small number of remaining locations, the commercial exploitation of *O. edulis* has almost ceased in Scotland. However, harvesting in the form of illegal gathering is still thought to be occurring at several locations within Scotland and represents a significant pressure on the population.

Additional threats to the population of *O. edulis* come from pests such as the particularly harmful slipper limpet (*Crepidula fornicata*) and, to a much less extent, the introduced American whelk tingle (*Urosalpinx cinerea*). Declining water quality due to pollution also had a detrimental impact on *O. edulis* stocks. Tributyl tin anti-fouling paints used on ships and leisure craft in the early 1980s caused stunted growth and probably affected reproductive capacity (Laing *et al.*, 2006). Finally, the current burden on introduced pathogens into the UK, e.g. *Bonamia ostreae*, is probably the greatest threat to overall population recovery at this time. Unfortunately, the small size of many remaining individuals makes them more susceptible to the effects of variable recruitment, disease, adverse environmental conditions and predation. These Allee effects are discussed in more detail below.

### 3.8.2 Overview of relevant studies

Restocking for commercial reasons has been undertaken many times throughout Europe and the British Isles. The stated recovery point with these activities was typically to restore depleted fisheries to a point of commercial productivity. Recent attempts of *O. edulis* restoration in the UK are at their initial stages. A project to restore habitat and regenerate stocks in Strangford Lough involved extensive cultch deposition and restocking with seed and brood-stock oysters. Initial results were promising with an increase in intertidal *O. edulis* densities (Roberts *et al.*, 2005). Unfortunately the period of restocking and cultch provision was short-lived and subsequent surveys have reported a decline in *O. edulis* numbers which has been attributed to unregulated fishing activity.

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 also affected by a series of factors including disease and habitat change (Woolmer *et al.*, 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 *et al.*, 2011). The Danish *O. edulis* restoration programme was one of the few studies to have observed some success from a fishery perspective with landings having increased from virtually nil in the year 2000 to over 900 tonnes in 2003.

There are also valuable lessons to be learned from the practical experience of an extensive native oyster restoration programme in Chesapeake Bay. These efforts have generated a substantial body of literature and reviewed in Rothschild *et al.*, (1994), Mann and Powell (2007) and Kennedy *et al.* (2011).

### 3.8.3 Factors influencing recovery

Features that may limit recoverability in *Ostrea edulis* include 1) brood stock size, 2) decrease in preferred settlement surfaces (gregarious behaviour), and 3) adult mortality (disease and harvesting).

#### 3.8.3.1 Brood stock size

As a species with a long history of commercial exploitation, there has been a dramatic reduction in the population of *O. edulis* in the UK since the middle of the last century (Laing *et al.*, 2006). Along North American, European and Australian coastlines, native oyster populations have been reduced to less than 10% of their historical abundance through overfishing and oyster reef destruction (Schulte *et al.*, 2009). The decline within the UK, especially on the North Sea coasts, can also be attributed to unusually cold winters in the 1930s and 1940s (Crisp, 1964 in Laing *et al.*, 2006).

As with any restoration effort, brood stock is required to provide the progeny for recruitment back into the adult population. Substantial amounts of gametes must be produced to overcome the losses associated with external sperm release and larviparous development, veliger mortality in the water column (whether from predation or dispersal into hostile habitats), unsuccessful settlement and juvenile mortality. Using the ecological definition of recruitment being juveniles becoming sexually mature, it is apparent that the brood-stock must generate vast amounts of gametes to overcome these numerous losses that lead up to recruitment into the adult population. Processes that lead to the failure of greatly reduced populations, such as the failure to provide enough gametes to sustain the population, are collectively termed Allee effects.

Synchronised spawning events, the retention of eggs by females prior to fertilisation with sperm that has been collected from the external environment and the release of gametes as concentrated packets, have been suggested as adaptations that overcome the inefficiencies of external fertilisation in the marine environment. These are all characteristics of the reproductive behaviour of *O. edulis* and may increase the effective distance over which sperm can fertilise eggs and provide an advantage when individuals exist at low population density (University Marine Biological Station Millport, 2007). However, population density and abundance also have a significant influence on the fertilisation success of individuals.

With regard to these Allee effects in *O. edulis* populations, Kennedy and Roberts (2006) studied the greatly diminished population in Strangford Lough, Northern Ireland. They suggest that Allee effects were probably responsible for preventing the recovery of this population and stated that the oyster population, and associated total reproductive effort, had probably fallen beneath a required threshold to offset the heavy losses associated with a planktonic recruitment phase. Fertilisation rates may have also decreased as the oyster population declined as the mean communication distance between compatible individuals' increased and rising dilution factors diminish stochastic elements required for fertilisation (Rothschild *et al.* 1994). Experiments undertaken by University Marine Biological Station Millport (2007) examined sources of potential Allee effects at two oyster sites within Loch Sween (Scotland). Characteristics of the *O. edulis* populations at the two sites included a fluctuating sex ratio, low levels of population aggregation and areas of very low overall site densities, suggesting that Allee effects could be important in limiting recruitment success within the *O. edulis* population examined.

Small adult populations are also disproportionately at risk from stochastic disturbance events and human impacts. Mortalities inflicted on these populations are proportionally greater than that suffered by larger and more widely distributed populations. Furthermore, selective removal of sexually mature oysters by unlawful gathering can increase the potential of Allee effects by further decreasing the population density and abundance.

The reduction in the adult population size also has important repercussions for the availability of optimum settlement substrata for the larvae (see below). This issue is particularly important for gregarious species such as *O. edulis* when the adult population size is proportional to the propagule pressure as well as the availability of preferred settlement surfaces.

### 3.8.3.2 Decrease in the availability of preferred settlement surfaces

Like other bivalve species that settle gregariously, oysters rely on adults for the production of larvae and for providing the preferred surfaces for settlement and successful recruitment into the adult population. *Ostrea edulis* larvae are pelagic for a period of 7 to 12 days, after which settlement occurs with larvae attaching to solid substrata and metamorphosing into spat (juvenile oysters). Key factors determining recruitment success in oyster populations are the number of larvae retained within the area of the oyster bed that successfully settle and metamorphose, and the availability of suitable substratum for larval attachment (Mann and Powell 2007).

*Ostrea edulis* will settle on many hard surfaces, e.g. compacted silt, muddy gravel with shells, sand and rocks (Laing *et al.*, 2005) although the species showed a clear preference for shell material over gravel or pebbles where survival is higher (UMBS, 2007 in Shelmerdine and Leslie, 2009). Most oyster species show gregarious settlement behaviour and preference for conspecific shell material. The growth rim of the adult *O. edulis* has been reported by Korringa (1941) to provide the best setting surface for recruiting spat of this species.

Although existing conspecific adults are the best substratum for settlement, dead shell is also a suitable substratum for settlement. This has been known by shellfish managers for centuries who routinely use dead shell, typically from the target species (cultch) to provide settlement surfaces (Kennedy and Roberts, 2006). However, Mann and Powell (2007) point out that cultch and existing shell from wild populations are resources that deteriorate constantly through taphonomic processes (burial and dissolution). Mann and Powell (2007) argue that any oyster restoration (*Crassostrea virginica* in this example) must deal with two interrelated issues. Firstly, a large brood stock must be amassing to provide larvae for recruitment and secondly that the production rate of settlement surfaces, i.e. live and dead shell, exceeds the rate of decay for this substratum. With regard to the second requirement, it is therefore important to note that healthy beds of oysters are not just reliant on recruitment for long term success but also on a sustainable amount of mortality. This mortality provides the habitat with the required production of shell that sustains the oyster habitat and provides the required surfaces for settlement. This fundamental need for mortality in *C. virginica* populations provides an additional burden on the rate of recruitment, and hence size of the brood-stock (Mann and Powell, 2007). A 57 fold increase in abundance of oysters (peaking at 185 million oysters within the site) was observed when these two factors were combined within a site closed to oyster harvesting in Chesapeake Bay, North America (Schulte *et al.*, 2009). For this species, natural recovery was observed to be exceptionally high given the right conditions.

Although *C. virginica* is a reef forming species, the same concepts apply to *O. edulis* populations in the UK. Without these preferred substrata for settlement, larvae will settle on other less suitable surfaces which are associated with higher levels of mortality (UMBS,

2007 in Shelmerdine and Leslie, 2009), hence weakening the restoration efforts. Equally, larvae settling in poorer habitat may also experience slower growth, poorer condition and ultimately reduced fecundity which again undermines recoverability.

The University Marine Biological Station Millport (2007) examined the availability of settlement surfaces at several sites in Scotland. Although *O. edulis* were recorded attached to all the differing types of substrata present at the survey sites, the results did not demonstrate total availability of substratum was a limiting factor to population growth. There was evidence that an increase in the availability of shell could increase larval settlement in these populations. The patchy nature of this preferred substratum when compared with the greater availability of sand and homogeneous settlement patterns suggest that if the availability of conspecific shell deposits were increased, settlement by larval oysters may increase. The authors ultimately concluded that the total availability of substrata suitable for settlement is quite high but the patchy nature of its spatial distribution is potentially causing a demographic bottleneck.

### 3.8.3.3 Adult mortality (disease and harvesting)

Although *O. edulis* is susceptible to several sources of natural mortality, the current burden placed upon this species by introduced diseases and pests is particularly great. The most potent source of morbidity in *O. edulis* is currently from *Bonamia ostreae* which causes Bonamiasis. The disease spreads through the movement of infected stock and now occurs in most of the major oyster producing areas (Laing *et al.*, 2006). Significant mortalities of animals over three years old are correlated with increased summer water temperatures and mortality rates in excess of 80% have been noted (Laing *et al.*, 2006). The effect this can have on yields can be seen in the drastic (93%) drop in recorded production in France, from 20000 tonnes per year in the early 1970s to 1400 tonnes in 1982 (Lapegue *et al.* 2004 in Laing *et al.*, 2006). Marteiliosis (*Marteilia refringens* – first reported August 2011), Bonamiasis (*B. ostreae* and *B. exitiosa* – latter first reported December 2010) and oyster herpesvirus (OsHV-1 – first reported August 2012) are also present in the UK, although as of 2009, *B. exitiosa*, *M. refringens* and OsHV-1 have not been found in Scotland (Shelmerdine and Leslie 2009).

The combined disease-related mortality for *O. edulis* is clearly significant and is widely recognised as a bottleneck for population growth. The elevated mortality of 80% for individuals over three years old is significant in a population where average life expectancy is between 2 – 6 years, and potentially over 10 years. Furthermore, age, body size and total fecundity are typically related in bivalves. A disease that ultimately generates mortality rates near 80% for the young adults will have implications for the overall age structure and ultimately reproductive output of the population.

Newly introduced pests and predators are also an important source of mortality for *O. edulis*. The main pest species is *Crepidula fornicata* that can become super-abundant on oyster grounds. Blanchard (1997) suggests that *C. fornicata*, exclude oysters in three ways, 1) domination of the substratum which prevents the successful recruitment of oyster larvae, 2) modification of the bathymetry through massive quantities of biodeposition and disruption of near-bed hydrodynamics that favour *C. fornicata* and 3) competitive dominance for suspended food mediated through high abundances of *C. fornicata*. These three factors reduce the growth and reproductive output for oysters present with *C. fornicata*, and exclude settlement for larvae attempting to recruit onto infested ground. With regard to predation from introduced species, the American whelk tingle (*Urosalpinx cinerea*) has been responsible for heavy mortalities of *O. edulis* in certain southern oyster beds (Woolmer *et al.*, 2011).

The other obvious source of adult mortality is the targeted extraction of individuals during fishing and unlawful gathering. Most of the current production of *O. edulis* is from aquaculture and is therefore independent of the wild stock. An interesting interaction between aquaculture and wild populations of oysters was suggested by Kennedy and Roberts (2006) that transient populations may actually represent significant brood stocks for wild beds. With regarding to harvesting, Smyth *et al.* (2009) observed that the *O. edulis* population within Strangford Lough, Northern Ireland, declined from an estimated 1.2 million in 2003 to 650,000 by 2005. This decrease of 45% of the population is attributed to unregulated harvesting of wild intertidal individuals. As stated above, when adult population sizes become very small, they are disproportionately affected by natural and anthropogenic impact, e.g. illegal harvesting.

#### **Box 9. Conclusions regarding therecovery potential of *Ostrea edulis***

Of the components of recoverability, three factors have been identified that limit this species, 1) diminished brood stock, 2) shortages of high value settlement surfaces, and 3) high adult mortality (disease, pests and targeted extraction). There have been several efforts to reintroduce *O. edulis* in various areas of the UK with varying levels of success. Based on the amount known about these specific bottlenecks for this species, the willingness of public and private sectors to the management and recovery of this species and its availability for translocation from other areas, the overall potential for recovery of this species appears high.

Established restoration techniques are tried and tested for each of the three bottlenecks and are sufficiently advanced so that they can be applied with some certainty of success. The time-span for recovery will depend on (i) the availability of spat or translocated brood-stock, (ii) the extent of the restoration area, (iii) the required restoration density (and additional biodiversity indicators) specified by the recovery reference range and (iv) the intensity of processes influencing growth, reproduction and mortality at the site. Given ideal conditions, site-scale recovery should be possible 6 – 12 years.

### **3.9 Seagrasses (*Zostera* species)**

Seagrasses provide an important food resource, habitat and nursery area for fish, seahorses and pipefish and play a role in maintaining sediment stability. *Zostera* species (*Z. noltii*, *Z. marina*, *Z. angustifolia*) generally occur in intertidal and lower shore to shallow subtidal (*Z. marina*, *Z. angustifolia*) muddy sands in full and variable salinity.

#### **3.9.1 Threats and sensitivity**

*Zostera* species (and other seagrasses) are highly sensitive to physical disturbance in the form of abrasion (surface and sub surface), physical removal, loss or change to the habitat. Additionally, seagrasses are highly sensitive to changes in water clarity, siltation rate changes and organic enrichment. Disturbance in seagrass beds can occur at a localised scale (grazing, trampling, localised discharges, moorings, dredging, beach nourishment and construction work, which can lead to changes in sediment dynamics and water clarity) or can occur at much larger scales as a result of flooding and climatic events, leading to widespread changes in sediment dynamics and water clarity (Campbell and McKenzie, 2004). It is of note that recovery potential following natural, physical disturbances is considered to be greater than that following anthropogenic disturbance and most human activities resulting in sediment deposition are thought to cause permanent changes to the sediment and permanent negative effects on seagrass (Cabaço *et al.*, 2008). FEAST (Scottish Government, 2013) indicated moderate sensitivity to changes in tidal currents, wave exposure and nutrient enrichment. However, eutrophication, resulting from nutrient enrichment, is considered to be a major cause of loss in many parts of the world (Cabaço *et*

*al.*, 2008). Land use changes in coastal areas have exacerbated problems such as increased sediment and nutrient load (Thrush *et al.*, 2003) which in turn, exacerbate the impacts of high turbidity, sediment dynamics and eutrophication on seagrasses (Campbell and McKenzie, 2004).

### 3.9.2 Overview of relevant studies

Whilst variable, generally according to the parameters used to determine recovery, *Zostera* species generally show 'potential' to recover following pressure removal. However, recovery times are generally long and recovery is not always successful (Mazik and Smyth, 2013) and in areas where widespread total loss has occurred, full recovery may be unlikely (Giesen *et al.*, 1990). Given the sensitivity of *Zostera* to nutrient and organic enrichment, and that these substances accumulate in the sediments (acting as a residual pressure), there may be a delay between management measures to remove or reduce pressures and the onset of the recovery process where physical recovery of the sediment may be necessary. This also applies to pressures which have led to erosion, deposition, changes to the textural characteristics of the sediment, compaction or liquefaction.

Documented recovery times for *Zostera* (in terms of recovery of shoot density and biomass) range from 2 to 7 years. For example, Campbell and McKenzie (2004) reported the disappearance of approximately 90% of intertidal seagrass following a large-scale flood in Queensland, Australia. Seagrass loss was attributed to increased turbidity, increased organic load in the water column and the sediment and increased bacterial oxygen demand within the sediment. Seedling growth was reported at 18 months with full recovery of the pre-flood spatial extent and shoot density being achieved within 2-2.5 years. Seagrass recovery was thought to be related to recovery of the sediment in terms of particle size distribution and organic content and a reduction in the concentration of suspended solids in the water column. It should be noted that this study was carried out in a sub-tropical habitat and in temperate areas, so seasonality could strongly influence recovery, in particular, growth rate and the duration of the growing season. It should also be noted that similar recovery times (24-30 months) have also been reported following small-scale (4 m<sup>2</sup>) experimental removal of *Zostera* with recolonisation occurring via lateral growth of the surrounding plants (Boese *et al.*, 2009). As such, recovery was slowest in the centre of the disturbed plots. These authors concluded that recolonisation was exclusively due to rhizome growth from adjacent vegetated areas (confirmed by poor survival of transplanted seedlings) and suggested that the timescale for recovery would increase with increasing scale of disturbance. Furthermore, Valdemarsen *et al.* (2011) found that the presence of bioturbators such as *Arenicola marina* could prolong the recovery process through seed and shoot burial to depths which prevent survival.

However, following more severe physical disturbance, for example that associated with mussel harvesting (dredging), recovery times may be considerably longer (Neckles *et al.*, 2005). In impacted areas, Neckles *et al.* (2005) found that recovery of between 1 and 61% of the pre-disturbance biomass had been achieved and that biomass remained substantially lower after 7 years. The predicted timescale for complete recovery was between 10 and 20 years. Cabaço and Santos (2007) documented negative effects of beach recharge on subtidal seagrass beds (*Posidonia oceanica*) and found that even if shoot density recovered, the proportion of live shoots had not. This was explained by a reduction in the amount of non-structural starch in the rhizomes which indicates mobilisation of carbon to meet the plants energy demands in the absence of light (an impact of the beach recharge).

### 3.9.3 Factors influencing recovery

The recovery potential of *Zostera* spp. is largely related to the nature and the scale of the disturbance that led to the decline. Recovery has generally followed direct pressure removal

and recovery of the physico-chemical nature of the habitat (e.g. reduction in suspended solids and/or nutrient concentrations, recovery of the physical structure of the sediment). The timescale for recovery increases with increasing scale of disturbance and with increasing homogenisation of the habitat. Recolonisation of *Zostera* is exclusively by rhizome growth from adjacent vegetated areas and therefore recovery potential is reduced as the degree of *Zostera* removal increases. Furthermore, permanent changes to the physical structure of the sediments may prevent recolonisation.

#### **Box 10. Conclusions regarding the recovery potential of *Zostera* species**

There is evidence that seagrass (*Zostera* species) can recover but recovery times may be long and, depending on the nature and scale of the disturbance, recovery may not be achievable in some cases. Seagrasses are highly susceptible to changes in nutrient status, turbidity and physical damage and documented recovery rates range from 2 years to over 7 years to reach a recovered state (Mazik and Smyth, 2013). Functional attributes, such as leaf and rhizome production and starch concentration can take more than 18 years to reach a recovered state, yet these attributes may be essential to long-term sustainability, and if the physical structure of the sediment is permanently changed, recovery may never be achieved.

### **3.10 Sandeels (*Ammodytes marinus* and *Ammodytes tobianus*)**

The ecological significance of sandeels lies in their importance in the food web, providing a staple food source for many species of fish, mammals and birds, particularly during the breeding season for birds (Harris and Wanless, 1991; Wanless *et al.*, 1998; Furness and Tasker, 2000). Sandeel species of interest as protected features of MPAs are *Ammodytes marinus* and *Ammodytes tobianus*. These two species are abundant in all UK waters. Sandeels live buried in the sediment, to depth of 20-50 cm (Heath *et al.*, 2012) and show a preference for sediments composed of fine sand and coarse, medium and fine silt (Holland *et al.*, 2005). *A. tobianus* is present from the intertidal to depths of approximately 30 m (Rowley and Wilding, 2008).

#### **3.10.1 Threats and sensitivity**

In the northern part of the North Sea, yearly landings have shown a large decrease since 2003 (ICES, 2006). In Norwegian waters, landings decreased by almost 90% in 2003-04 compared to 1994-02 (ICES, 2006). However, population density is highly variable between years. Sandeels are commercially fished for fish oil and fishmeal and in the Dogger Bank area of the North Sea, there is evidence of fishing-induced depletion with subsequent impacts on their predators (Engelhard *et al.*, 2008). Given the specificity of their preferred habitat, any form of disturbance that is likely to disrupt the physical structure of the sediment poses an indirect threat to sandeel populations.

According to FEAST (Scottish Government, 2013), sandeels in Scottish waters are highly sensitive to demersal trawl gear used for specifically targeting sandeels as it can cause local depletion and alter the age and size composition of the sandeel population. Additionally the species are also sensitive to activities that can cause physical damage to the sea bed (change, abrasion, loss, extraction) and siltation rate changes, such as renewable energy development, aggregate extraction, infrastructure development, military activities and coastal defence. Sandeels are moderately sensitive to organic enrichment and low levels of siltation associated with sewage/industrial inputs and aquaculture. High sensitivity to temperature is assumed although there is no indication of the level of confidence associated with this assumption (Scottish Government, 2013).

Predation from piscivorous fish (such as cod, haddock, whiting, saithe and mackerel), as well as some seabirds (in particular Atlantic puffin) occurs during the sandeel's first summer (age 0), whilst other avian predators (such as guillemot, razorbill, kittiwake and tern) and marine mammals prey on those that are age 1 or older (Greenstreet *et al.*, 2006; 2007). Between 1990 and 2009, sandeel biomass was reported to be unstable on the east coast of Scotland (ICES, 2009), with several studies being unable to explain the fluctuation in biomass. Demographic survey data show that despite the fluctuation in stock size, recruitment remained stable, and therefore stock decline was not due to recruitment failure (Heath *et al.*, 2012).

Van der Kooij *et al.* (2008) highlighted the sensitivity of sandeels to environmental variation, including factors such as temperature, light and salinity. Jensen and Christensen (2008) also suggested that food availability may be an important factor, reporting a decrease in sandeel abundance in line with a decrease in zooplankton abundance leading to a reduction in the number of juvenile sandeels reaching sexual maturity. It has also been shown that climatic variability affects the population dynamics of the zooplankton food of sandeels (Malzahn *et al.*, 2007). Arnott and Ruxton (2002) demonstrated a negative relationship between recruitment and the winter index of the North Atlantic Oscillation, which affects sea temperatures during the spawning and larval period. In addition, warmer sea temperatures were shown to correlate with poorer than average recruitment, suggesting climate change could influence sandeel populations in the North Sea (Arnott and Ruxton, 2002).

### 3.10.2 Overview of relevant studies

Following closure of the sandeel fishery the north-western North Sea (Forth region) in 2000, Greenstreet *et al.* (2010) documented an initial and immediate marked increase in sandeel biomass. This was immediately followed by a steady decline. There was an indication of the biomass stabilising in 2005-06, at a level higher than that of 1997-99 when the fishery was operational, but 2007 saw the biomass decline again, to levels similar to those in 1997-99. Biomass was still low in 2008, but again showed evidence of recovery in 2009, likely related to a stronger juvenile group than the two previous years (Greenstreet *et al.*, 2010). Results from monitoring the piscivorous fish showed no correlation between their biomass and the biomass of their sandeel prey. Of the six seabirds monitored, gannets and shags appeared unaffected by changes to sandeel biomass, whilst numbers of kittiwakes and three auk species increased at the time of fishery closure, before declining again as sandeel recruitment diminished.

### 3.10.3 Factors influencing recovery

Features that may limit recoverability of sandeels include climatic variability (particularly water temperature) and its influence on egg and larval development and subsequent recruitment and the abundance/dynamics of zooplankton. The discrete timing of spawning and hatching in many temperate fish species, such as sandeels, is believed to be adaptive, reflecting the environmental conditions most favourable by that species for egg and larval survival. This means they have highly variable recruitment dynamics. Recovery and maintenance of suitable substratum is of primary importance, given the habitat specificity of sandeels and the requirement of suitable substrata to support egg development (Rowley and Wilding, 2008). High levels of silt are thought to impede respiration and it is of note that *A. marinus* has been reported to be absent from sediments with a silt/clay content of >10% and densities decreased between fractions from 2 to 10% (Wright *et al.*, 2000).

The fluctuation observed by Greenstreet *et al.* (2010) in the north-western North Sea (Forth region) suggested that the local abundance of sandeels is strongly dependent on recruitment, which is subject to environmental influence. Results from monitoring the

piscivorous fish showed no correlation between their biomass and the biomass of their sandeel prey.

**Box 11. Conclusions regarding the recovery potential of sandeels (*Ammodytes marinus* and *Ammodytes tobianus*)**

Overall, population dynamics of *A. marinus* appear to be driven by significant changes in the highly variable recruitment dynamics of this species. The recruitment of *A. marinus* is driven predominantly by environmental factors. With regard to recoverability of this species, many of the driving factors are beyond control of managers, e.g. oceanographic factors. However, the requirement for clean substrata does provide an avenue for influencing recoverability, either through the greater protection of these seabed areas or active restoration of them.

The timescale for recovery will be dependent on (i) the size of the remaining population within the recovery area, (ii) the rates of recruitment and mortality, (iii) the population size used for the recovery reference range and (iv) the rate of dispersal of sandeels outside the restoration area. Given ideal conditions, noticeable changes in population size could be apparent after 4-6 years (assuming 50 % of the individuals are sexually mature after 2 years and therefore recruitment from the follow cohort will recruit after another 2 years). The overall timescale to reach a recovered state will depend on the size and dynamics of particular populations however it is anticipated that many generations will be required, i.e. timescales greater than 4 - 6 years.

### **3.11 Common skate (*Dipturus cf. flossada* and *Dipturus cf. intermedia*)**

The “common skate species complex” refers to the species formerly known as *Dipturus batis*, and is recognised as a complex of two species provisionally *D. cf. flossada* and *D. cf. intermedia* (Iglesias *et al.*, 2010). Common skate live on sand and mud substrata, with adults inhabiting depths of 10 to 600 m and younger individuals inhabiting shallower waters (Neal *et al.*, 2008).

#### **3.11.1 Threats and sensitivity**

Commercial fishing has brought the common skate to the edge of extinction and the species was upgraded to Critically Endangered on the 2006 IUCN Red List of Threatened Species (Iglesias *et al.*, 2010; iucnredlist.org). As such, landing of common skate is now illegal and by-catch must be returned to the sea. In the UK, skate landings were all reported as ‘skates and rays’ regardless of species until 2008-09, which lead to a lack of species specific monitoring on their relative abundances (ICES, 2012). Walker and Hislop (1998) showed that common skate was abundant between 1929-56, but by 1981-95 it was no longer being caught by fishing vessels. Skate size also decreased over this period. The common skate was previously identified as the most commercially important of the five species of skate but it is also the species whose life history characteristics show most sensitivity to change and enhanced mortality (i.e. long life-span and late maturation) (Walker and Hislop, 1998).

Common skate shows moderate sensitivity to collision (death or injury caused by interaction with sea bed structures e.g. tidal turbines), surface abrasion and siltation rate changes (FEAST, Scottish Government, 2013) which can affect egg cases in particular laid on the seabed. Activities giving rise to these pressures include renewable energy development, aquaculture, demersal fishing activities, military activities, dredging, and aggregate extraction.

Griffiths *et al.* (2010a), discovered that common skate existed as two cryptic species (i.e. indistinguishable morphology but belonging to different species which cannot interbreed) and

suggested that major declines had actually occurred in two species, with both being critically endangered. It was suggested that the distribution of these two cryptic species was limited by temperature, with association with other habitat variables being similar for both species. As conventional fisheries management is mainly based on the concept that a population is spatially and temporally homogenous owing to a common characteristic, knowledge that this species is not confined this way is important in terms of monitoring the species, and could prevent overexploitation of local stocks (Mucientes *et al.*, 2009).

### 3.11.2 Overview of relevant studies

As the population of common skate is currently identified as critically endangered, there is no evidence of populations recovering at this time. This is further complicated by the possible cryptic nature of the species, and incomplete knowledge of habitat and movement preferences off the coast of Scotland, where it is understood they are at their most abundant, as well as a lack of data on the location of spawning and nursery grounds. Therefore it is clear that more information is needed on the localised populations of common skate.

### 3.11.3 Factors influencing recovery

Features that may limit recoverability of common skate include 1) re-colonisation potential (high level of site fidelity and clade/population geographical separation), and 2) slow recruitment due to life-history characteristics (long life-span and late maturation)

Longevity and low fecundity dictate slow and uncertain recovery times for this species. Neal *et al.* (2008) indicate that common skate can live for 20-100 years and do not reach sexual maturity until about 11 years. Additionally, females only produce up to 100 eggs (with a high mortality rate) and spawning only occurs every second year (Neal *et al.*, 2008).

Common skate are generally assumed to have a sedentary lifestyle, spending much time on the sea bed, generally at the same site. Little (1998) observed a high level of site fidelity for highly localised sites, with only 3% of tagged and released skate moving more than 20 km from the release site, even after long periods of time. Within these localised areas however, time-depth profiles showed periods of high activity, including vertical movement of over 100 m conducted on a regular, sometimes daily basis, although smaller skate were generally less active and resided in deeper waters. The limited horizontal movements in preferred habitats suggests high foraging activity in the water column, and the need for spatial management of localised populations of this once widespread fish (Wearmouth and Sims, 2009). This lack of mobility is likely to limit interaction between populations and is likely to significantly limit recovery potential. Furthermore, in the longnose skate *Dipturus oxyrinchus*, Griffiths *et al.* (2010b) reported genetic isolation and limited connectivity between populations and suggested that this, together with low fecundity and low dispersal potential, could limit the recovery potential of this species. Ellis *et al.* (2011) highlighted the lack of data on the occurrence and distribution of egg cases of common skate, which increases the difficulty associated with effective conservation of their spawning grounds.

**Box 12. Conclusions regarding the recovery potential of common skate (*Dipturus cf. flossada* and *Dipturus cf. intermedia*)**

As the population of common skate is currently identified as critically endangered, there is no evidence of populations recovering at this time. Based on the small size of the current population and low recruitment (typically of a large, long-lived species), the estimated rate of recovery for common skate is considered to be very slow. The factors influencing the actual timescale required for recovery will include (i) the size of the remaining population, (ii) the rates of recruitment and mortality (mainly from fishing activity), (iii) the population size used for the recovery reference range and (iv) the rate of dispersal of skate outside the restoration area. As such, noticeable changes in population size are likely to require decades and reaching a recovered reference range from a current population level is likely to require timescales greater than 50 years. This is further complicated by the possible cryptic nature of the species, and incomplete knowledge of habitat and movement preferences off the west coast of Scotland, where it is understood they are at their most abundant, as well as a lack of data on the location of spawning and nursery grounds. No active restoration attempts have been attempted and existing methods such as nursery techniques appear generally to be inappropriate.

### 3.12 Herring spawning ground

The Atlantic herring (*Clupea harengus*) consists of both winter-spring and summer-autumn spawning groups, characterised in the north-east Atlantic by oceanic, shelf, and coastal populations. Oceanic groups are large migratory fish that spawn off the Norwegian and Icelandic coastline, the shelf group consists of migratory North Sea populations adjacent to the British Isles, and the coastal groups are generally smaller fish restricted to the Baltic and White seas (Haegele and Schweigert, 1985). *Clupea harengus* reach maturity between 3 and 9 years and feed predominantly on small shrimps or copepods but have been known to filter-feed in particular circumstances (Barnes, 2008).

*Clupea harengus* is a pelagic species that occurs in surface waters down to a depth of around 200 m (Barnes, 2008). Outside of the spawning season, *C. harengus* reside offshore and are often found in large near-surface shoals (Barnes, 2008). Herring off the coast of Scotland are considered to be conservative with respect to their spawning grounds, which are usually located in high energy environments and include both nearshore and tidally active positions depending on whether they are spring or autumn spawners (Blaxter and Holliday, 1963). Spawn is deposited on marine vegetation or on predominantly gravelly substrata, which is free from silting (to avoid egg mortality from suffocation). Parrish and Saville (1965), Blaxter (1990) and Haegele and Schweigert (1985) showed herring to have a preference for spawning on high energy gravel substrata.

Sediment type, bottom depth and prey abundance appear to be key determinants of North Sea herring distributions (Maravelias *et al.* 2000). Prior to spawning, herring have been shown to aggregate in areas characterised by sand and gravel sea bed, which is similar to their spawning ground preferences (Haegele and Schweigert, 1985). The herring additionally showed a preference for zooplankton rich waters within a depth range of 100-150 m, suggesting a relationship between spatial distribution patterns and environmental factors (Maravelias *et al.*, 2000).

#### 3.12.1 Threats and sensitivity

Given the specificity of their preferred spawning ground, herring are potentially highly susceptible to physical disturbance, particularly that which results in substratum modification or siltation. Activities likely to impact upon the physical integrity of the substratum include aggregate extraction (de Groot, 1980), particularly where screening is involved (where the

fine fractions are returned to the sea resulting in a change from, for example, a gravelly to a sandy bed), dredging and the deposition of material onto the sea bed. Oulasvirta and Lehtonen (1988) found no direct impact of gravel extraction on herring spawning (although extraction was taking place 1.5 km or more from the spawning sites). They did, however, observe a decrease in herring catch during dredging, possibly due to avoidance of the associated noise by the herring. It is emphasised that the effects of gravel extraction on herring spawning grounds will be specific to the nature of the substratum, the dredging practice, the proximity of the spawning grounds to the activity and the local hydrodynamic conditions which may or may not transport fine, suspended sediment to the spawning grounds. It is also emphasised that aggregate extraction and screening practices may not be relevant to Scottish waters, but the documented physical effects of these activities may be similar to those associated with other activities exerting direct, physical pressures on the sea bed. For example, fishing techniques which result in significant sediment resuspension or the passage of fishing gear directly over spawning grounds (particularly when eggs are present) present a potential threat, as does physical disturbance associated with infrastructure and offshore energy development. Morrison *et al.* (1991) reported mass mortality of herring eggs following a diatom bloom.

### 3.12.2 Overview of relevant studies

Physical recovery of degraded herring spawning grounds is essential to ensure successful spawning, recruitment and population sustainability. Boyd *et al.* (2004) compared dredging at several east coast sites in the UK, and found that the intensity of the dredging activity influences the rate of recovery for the physical characteristics of the substratum, as well as its benthic community once the dredging has ended. In terms of the physical recovery of the sea bed scarring/ tracks were still evident 3-10 years after the cessation of activity (Boyd *et al.* 2004). However, whether or not the remaining habitat was suitable for spawning is dependent on the sediment structure and particle size distribution, its stability and the potential for recolonisation of algae to which herring may attach their eggs.

### 3.12.3 Factors influencing recovery

Information on the recovery of herring spawning grounds, as a habitat, was not found. However, studies on the recovery of coarse sediment habitats in general may provide an indication of potential recovery times. For example, following aggregate extraction, Foden *et al.* (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. This indicates that not all aspects of physical recovery are necessary for the habitat to support a particular community type (or provide suitable habitat for herring to spawn). For example, one of the criteria used by Hall-Spencer and Moore (2000b) and Foden *et al.* (2009) was the disappearance of dredge tracks but the importance of this topographic change to biological community development is not clear.

Assuming that the physical structure of herring spawning grounds can recover, features that may limit recovery of *Clupea harengus* include; 1) the availability of spawning habitat (marine vegetation or on predominantly gravel substrata), 2) egg mortality (siltation suffocation, direct predation, high densities spawning and habitat disturbance by wave action and air exposure in nearshore areas), and, 3) noise (e.g. dredging, sediment extraction, construction) may drive schools of herring away from spawning areas.

With respect to the fish, Schmidt *et al.* (2009) hypothesised that as herring stocks collapse, the diversity of spawning sites also collapses, which was found to be the case in North Sea autumn-spawning herring, which collapsed in the late 1970s. As the recovery of the stock

took place (in terms of biomass), re-colonisation of old spawning sites also occurred. After time, from collapse to recovery, there was almost no change in the number of spawning locations where the highest abundances of larvae (top 50%) were found (Saville and Bailey, 1980).

The number of core sites was maintained throughout collapse and recovery, but the distribution of sites was different, perhaps due to additional environmental factors. Although the reasons behind re-colonisation of certain spawning grounds are unclear, this research suggests that it may be due to migrants “re-discovering” or “re-learning” migration routes. It should be noted that in some areas, re-colonisation took approximately 25 years and therefore conservation measures for herring spawning grounds may not have an immediate effect, and monitoring of each spawning component is important (Schmidt *et al.*, 2009). A similar collapse in the Georges Bank fishery in the Maine region, USA in 1976 was studied by Smith and Morse (1993). They charted how the fishery recovered between its collapse and 1990, and concluded that the rebuilding process was attributed to re-colonisation rather than resurgence, so the ability for herring to be able to return to once deserted spawning grounds is important (Smith and Morse, 1993). It is therefore important that the spawning grounds are available to the herring, and free from disturbance such as dredging or aggregate extraction.

#### **Box 13. Conclusions regarding the recovery potential of herring spawning ground**

Based on studies of physical recovery of the sea bed following gravel extraction, it is likely that herring spawning grounds have the potential to recover. However, the timescale is unclear. Documented physical times to achieve a recovered state range from 5 to 20 years but a biological recovered state has been reported within 5-12 years. The timescale for recovery of the biological components of the habitat which are of importance to herring is not known.

Regarding herring, population-level recovery has been observed for this species along with the re-colonisation of old spawning sites. As the remaining pressures of sedimentation and noise can be managed at source, it would seem that the overall recoverability of this species is high. The application of active restoration methods have either not been tried or seem generally inappropriate. The recovery reported to-date appears to require pressure removal and the maintenance of the remaining spawning grounds.

### **3.13 Black guillemot (*Cephus grylle*)**

The black guillemot (*Cephus grylle*) is a mid-sized auk of the true guillemot family. Black guillemot is largely sedentary across its range, except in areas where coastal areas freeze (Butler and Buckley, 2002). Black guillemot differ in their life cycle and ecology from other species of auk breeding in Britain and Ireland, i.e. guillemot (*Uria aalge*), razorbill (*Alca torda*) and puffin (*Fratercula arctica*). Black guillemots lay two eggs as opposed to one allowing the species to respond more rapidly to population changes. Furthermore, the species has a very eclectic diet consisting of zooplankton, crustaceans, fish and molluscs, and appears capable of switching preferences according to local abundance (Ewins, 1990).

The black guillemot breeds away from the large seabird cliff colonies and prefers small rocky islands and low lying, indented stretches of rocky coast. As such, the species is dependent on rocky mainland coast and islands in Britain (Mitchell *et al.*, 2004). Black guillemot nests are generally hidden in rock crevices and under boulders. The species usually chooses natural holes and crevices which are not visible from the entrance (Harris and Birkhead, 1985). The species also uses man-made nest sites, including holes in harbour walls, under

floors and roofs in buildings, and nest-boxes (Greenwood, 2002). Again, the species chooses the least exposed and least disturbed cavities in artificial nest sites such as piers (Greenwood, 2002). The distribution of black guillemot in Britain and Ireland is controlled by the availability of suitable nesting sites, according to Mitchell *et al.* (2004). At sea, black guillemot are rarely seen more than 5km from land or in water greater than 40m deep (Ewins and Kirk, 1988). In Shetland, birds tend to forage where the sea bed is rocky and vegetated with dense stands of kelp (*Laminaria* spp.) (Ewins, 1990).

### 3.13.1 Threats and sensitivity

In the context of Scottish waters (based on FEAST, Scottish Government, 2013), the species is highly sensitive to the introduction of non-indigenous species and species translocations. The species sensitivity is also considered at a medium level for several pressures: barrier to species movement, death or injury by collision, underwater noise, particularly in relation to activities such as energy production and associated infrastructure at sea, fishing (excluding activities on intertidal areas) and military (FEAST, Scottish Government, 2013).

Because of their near shore distribution, black guillemots are considered to be very sensitive to adverse effects from tidal stream turbines and wave energy devices in the UK (Furness *et al.*, 2012). The species is also sensitive to disturbance from boat activity and human presence. Indeed, black guillemot feeding at sea can respond to disturbance created by boat activity, as shown in the vicinity of breeding colonies in the Bay of Fundy (Canada) (Ronconi and Clair, 2002). Human interferences at breeding sites can also lower breeding success. Cairns (1980) found the reproductive success of black guillemot nesting in the Gulf of St. Lawrence subjected to daily visits by a surveyor to be much lower than in sites visited once every four days.

The species is also sensitive to incidental mortality from oil spills and evidence of effects on populations is well documented, particularly in Shetland where major oil spills have occurred in 1978 and 1993 (Heubeck and Richardson, 1980; Heubeck *et al.*, 1993; Heubeck, 2000). Heubeck and Richardson (1980) reported that the 1978 bunker fuel oil spill at the Sullom Voe terminal in Shetland wiped out the Yell Sound population, with 729 black guillemot found oiled. In 1993, following the spill of 85,000 tonnes of crude oil at Garths Ness on the south-western tip of mainland Shetland, 219 oiled black guillemots were found but the total number reported to be killed was thought to be around 1,300 of which 84% were adults (Heubeck *et al.*, 1993). Heubeck (2000) also reported measurable declines in the population on south-east mainland Shetland following other minor oiling incidents occurring in Shetland between 1985 and 1991.

Black guillemots at breeding sites are also particularly vulnerable to mammalian predation whereby the presence of mammalian predators can reduce the maximum limit on population size at a colony. Furthermore, the introduction of predators to a previously predator-free colony may result in a decline in breeding numbers (Mitchell *et al.*, 2004). American mink, an introduced alien species in Europe, has induced changes in the nesting distribution of black guillemot in Iceland, whose largest nesting colonies are now concentrated on inaccessible offshore islands (Hersteinsson, 1999). Whilst there is limited published evidence on the impact of the mammalian predator presence at nest sites in Scotland, there is evidence to suggest that black guillemots return to breed at sites following the eradication of the predators, in particular rats (Zonfrillo and Nogales, 1992; Mitchell *et al.*, 2004). There is also circumstantial evidence that the spread of American mink is affecting distribution and number of black guillemots, notably in Argyll and Bute where the disappearance of black guillemots in some offshore islands and along the mainland coast has been attributed to American mink predation (Craik, 1997; Mitchell *et al.*, 2004). Craik (1997) found that feral American mink caused widespread whole-colony breeding failures of black-headed gulls (*Chroicocephalus ridibundus*), common gulls (*Larus canus*) and common terns (*Sterna*

*hirundo*) at colonies on small islands in a study area along 1,000km of mainland coast in west Scotland. After one or more years of such failure, most of the affected breeding sites held no birds or greatly reduced numbers. Although there is no indication of the numbers of pairs of black guillemot affected in the study by Craik (1997), the author reported that black guillemot which also commonly bred on the small islands surveyed disappeared after mink predation. Otter (*Lutra lutra*) also have the ability to swim and visit offshore islands and the potential to affect breeding numbers but there is no evidence of otter-driven black guillemot declines in Scotland (Mitchell *et al.*, 2004).

The impacts of fisheries on seabirds are extensively documented in the literature. Impacts of fishing activities were reviewed by Tasker *et al.* (2000) and Furness (2003), based on case studies from all around the world, with particular focus on fisheries causing direct mortalities and changes of food supply. Concrete evidence of fishing activities impacting on black guillemot population is lacking in Scotland and further afield. Whilst it is recognized that black guillemot can become entangled in fishing nets (e.g. fixed gear type nets) and subsequently drowned, there is little published evidence to suggest a direct impact on populations; other than the analysis of ringing recovery in the UK which shows that the most important cause of ringing recovery is drowning in fishing nets (Okill, 2002). Okill (2002) also adds that the expansion of the fish-farming industry in much of the species' British range has led to increased mortality from this cause in the last 20 years. Whilst there is no evidence to support this claim, Okill (2002) states that the loss of birds in fishing nets can reduce local numbers and any expansion of inshore or unregulated fisheries could seriously affect nearby populations.

Beside death or injury caused by entanglements in fishing nets, the reduction of prey availability can affect seabird populations. As with other auk species, one of the black guillemot's favoured prey item in UK waters is sandeels (*Ammodytes* sp.). Sandeels are also the target of the largest single-species fishery in the North Sea. Historically, over-fishing of sandeels was considered to be the main cause of decline of auk species in the North Sea, but there is more recent evidence of a progressive increase in sea temperature affecting the availability of sandeels to seabirds (Frederiksen *et al.*, 2004; Wanless *et al.*, 2007).

### 3.13.2 Overview of relevant studies

Evidence for recovery of the number of pre-breeding individuals following a major oil spill is reported in Yell Sound (Shetland) by Heubeck (2000). Although pre-breeding numbers were not measured prior to the 1978 *Esso Bernicia* spill oil at the Sullom Voe Terminal, Heubeck (2000) inferred that the 155% increase in the numbers of pre-breeding black guillemot recorded between 1983 and 1998 probably represented recovery from the 1978 spill, following which 729 black guillemots were found oiled, mostly on beaches in Yell Sound (Heubeck and Richardson, 1980).

Further evidence for recovery is shown at other sites in Shetland following known localised mortality of black guillemot from oil pollution incidents. Decreases in the number of pre-breeding individuals recorded in 1993 in south-west mainland sites, on Fair Isle and possibly at a site in south-east mainland were recorded after the January 1993 *Braer* oil spill. Each of these decreases was followed by an increase (Heubeck, 2000). In the case of the *Braer* disaster, pre-spill surveys of black guillemots were available and the subsequent monitoring showed that recovery to pre-spill levels took 1-5 years (Heubeck, 2000). The measure of recovery in this instance is based on numbers of pre-breeding individuals. Birds were counted in the pre-breeding season (late March to early May), when adults spend the first hours of daylight close inshore, or ashore, displaying near prospective nest sites.

Despite high mortalities caused by oil spills in Shetland, the population (measured by numbers of pre-breeding individuals) has shown the capacity to recover quickly. It must be emphasised however that Shetland is the species' stronghold in Britain and it is possible that the speed of recovery following a major oil spill disaster may differ outside Shetland. For example, it is unclear whether birds breeding in more isolated and fragmented areas or on the edge of the distributional range would recover as quickly as the Shetland population.

There is anecdotal evidence of recovery following the eradication of mammalian predators from islands in the UK. It remains however difficult to quantify the extent of the recovery as, often, the exact size of the black guillemot breeding population prior to the eradication of mammalian predators is unknown. Zonfrillo and Nogales (1992) reported that the eradication of rats from Ailsa Craig (Firth of Clyde) in 1991 coincided with the return of breeding black guillemot the same year and by 2001 there were 12 pairs breeding. Mitchell *et al.* (2004) also reported that the first pair of black guillemot bred on Puffin Island (Gwynedd) in 2002, four years after rats were eradicated. The recovery, measured in this instance by the number of breeding pairs and presence/absence of breeding pairs, is dependent on the surplus of non-breeders elsewhere (due to shortage of nest sites) and rely on predators not returning to these islands, according to Mitchell *et al.* (2004).

Outside Scotland, the effect of feral mink removal on a range of birds has been studied between 1992 and 2001 on small Islands in the Baltic Sea (Nordstrom *et al.*, 2003). Breeding density of black guillemot on these Islands was calculated from the numbers of pairs, estimated by multiplying the total number of birds seen in the colony by 0.5. The study indicated that black guillemot, already extinct from one of the removal areas, returned to breed in 1994. A second removal area studied over a shorter period (i.e. 1998-2001) already held small numbers of breeding black guillemot but during the four years no notable changes in population size or distribution were found there.

### 3.13.3 Factors influencing recovery

Evidence of recovery remains largely anecdotal for black guillemot and the conditions for recovery are lacking in details in the evidence reviewed. However, it can be inferred that a multitude of factors relating to the black guillemot biology and ecology can affect the recoverability of numbers: 1) natal fidelity / sedentary life style, 2) Predation, and 3) clutch size. For the majority of evidence reviewed, it must be emphasised that the recovery is judged in terms of counts of individuals.

#### 3.13.3.1 Natal fidelity / sedentary life style

Most individuals do not move far from their natal colony. In an analysis of all ringing recoveries of dead birds, Okill (2002) found the median distance moved from the colony to be only 10.5km ( $P5-95 = 0-186$  km,  $n=134$ ). The relatively low dispersive behaviour of black guillemot could be one of the factors that affect the speed of recoverability. For example, it is possible that breeding sites at the edge of range or away from the main centre of distribution may take longer to be re-colonised following large-scale mortality e.g. oil spill. The speed at which the black guillemot population has recovered in Shetland following the *Eso Bernicia* and *Braer* disaster was rapid, but this has occurred in the species' stronghold where the population of black guillemot is thriving. Given the paucity of recovery evidence elsewhere in Scotland, it is difficult to draw comparison with areas supporting lower breeding densities.

#### 3.13.3.2 Predation

The distribution of black guillemot is controlled by the availability of suitable nesting sites. When suitable, the choice of where to place a nesting site is dictated largely by predator accessibility to the site (Ewins, 1989; Greenwood, 2002). Mitchell *et al.* (2004) state that the

eradication of predatory mammals will only lead to successful re-colonisation by susceptible black guillemots if there is no chance of the predators returning. Contrary to rats, American mink and otters are proficient swimmers and could visit offshore Islands, potentially limiting the number of available safe nest sites. In the case of American mink, the recovery following the eradication of predators from a small island would depend on the accessibility of the site by visiting predators from the mainland and neighbouring islands.

#### 3.13.3.3 Clutch size

Black guillemots lay two eggs as opposed to one in most Atlantic auk species. It has been suggested that this allows the species to respond rapidly to population changes, although there is no scientific evidence to support this statement.

#### **Box 14. Conclusions regarding the recovery potential of black guillemot**

Black guillemots have shown the capacity to recover quickly following oil spills, although the impacts from studies reviewed was localised and occurred in Shetland - the species' stronghold in the UK. Recovery away from the main centre of distribution could be, however, hampered by the relatively sedentary nature of black guillemot, with black guillemot undertaking little movements from their natal colony, thus reducing dispersal to more fragmented or isolated populations. It appeared that recovery following eradication of mammalian predators is also evident from the studies reviewed. One of key factors controlling population size and distribution is the availability of suitable nesting sites (natural holes or crevices in rock with no access to mammalian predators). Efforts to restore population levels needs therefore to consider nest site availability before any restoration program (e.g. eradication of predators). It must be, however, emphasised that this species uses man-made sites to nest and it seems that recoverability may be enhanced by the provision of artificial nesting sites (e.g. nest boxes).

## 4. DEVELOPMENT OF A FRAMEWORK FOR ASSESSING RECOVERY POTENTIAL

### 4.1 Overarching questions relating to the recovery process

Section 3 provided an overview of the major threats to and sensitivity of the various features and summarised the factors that may influence recovery for each species and habitat. Of the species and habitats reviewed in Section 3, there are a number for which either there analysis found no significant recovery or that the study documented recovery but lacked the required longevity to fully capture the return to a recovered state. This latter issue was often related to aspects of their biology significantly limit recovery potential. For other species, recovery potential is unknown because it has not been studied. However, aspects of their biology may provide some indication of recovery potential. The recovery potential of each species, community or habitat is summarised below although it is emphasised that this is based on a limited number of studies. It is also emphasised that, in most cases, recovery was the direct result of pressure removal. Native oyster was an exception where recovery followed active restoration.

#### Species/habitats with documented recovery potential include

- *Limaria hians* (10 - >100 years to reach a recovered state) (no assessment of associated community)
- *Zostera* spp. (2-7 years, up to 18 to reach a recovered state)
- *Ammodytes* (timescale unclear due to variability in population size. Likely to be tens of years or less to reach a recovered state)
- Black guillemot (approximately 5 years to reach a recovered state)
- Herring spawning ground (5-10 years for physical recovery of habitat, 25 years for recolonisation and to reach a recovered state)
- Native oyster (*Ostrea edulis*) (following active restoration potentially 6 – 12 years to reach a recovered state)

#### Species/habitats with insufficient information

- Maerl or coarse shell with burrowing cucumbers (recovery is limited by distribution of maerl or coarse shell gravel but there is a need to investigate the potential for recovery of *Neopentadactyla mixta* in gravel).
- Burrowed mud
- Shallow tide-swept coarse sands with burrowing bivalves (there is evidence that some species within this habitat may recover but this depends on current population status. The timescale to reach a recovered state is likely to be tens of years and there is no information on 'community' recovery potential).
- *Atrina fragilis* (there is some evidence of successful translocation but the long-term survival is not known.)

#### Species/habitats with no documented recovery include

- *Modiolus modiolus* (if recovery is possible, it is likely to take tens or hundreds of years to reach a recovered state)
- Maerl (if recovery is possible, it is likely to take hundreds of years to reach a recovered state)
- *Arctica islandica* (if recovery is possible, it is likely to take hundreds of years to reach a recovered state)
- Common skate (recovery period likely to be protracted but may be further limited by low population density and low fecundity – recovery greater than 50 years)

Information on the recovery potential of many of these species was lacking and therefore these factors are discussed, in detail, in a wider ecological context, drawing on examples from a variety of species and habitats in Section 5.

In summary, the ecological factors influencing recovery potential of the species considered in this document include:

- scale of the disturbance and degree of impact, accounting for spatial extent, disturbance type, intensity, frequency;
- physical recovery of the habitat;
- degree of habitat fragmentation, connectivity and homogenisation;
- hydrodynamic regime in relation to connectivity (physical and ecological, defined in 5.1.4)
- spatial extent, distribution and condition of remaining species, communities (brood stock) and habitats ;
- life history traits including reproductive strategy, dispersal strategy, growth, longevity, mobility;
- recruitment;
- interactions with other species;
- change in distribution or habitat use;
- natural population variability (including natural extinction and resource exploitation).

Influential factors relating to management are likely to include:

- poor knowledge of species biology, distribution, cause of impact and severity of impact;
- appropriateness of management measures;
- appropriate spatial scale of management;
- appropriate timing of management.

Whilst similarities were found between features, the processes leading to recovery are likely to be species, community, habitat and location specific and will be highly dependent on historical issues leading to their decline. However, the type of information vital for assessing both the decline of a feature and its potential for recovery is often common between species and habitats. From a management perspective, the recovery process is based on a set of high level questions considered useful to managers and the stakeholders to whom they have to communicate the reasons for recovery (Box 15).

#### **Box 15. Questions relating to the recovery process**

- 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?

## **4.2 Considerations for assessing recovery potential**

A set of high level, generic considerations, based on the questions in Box 15, are presented in Tables 2 to 4 and described briefly here, with greater details relating to the underlying ecological considerations being provided in Sections 3 and 5. Specific considerations for individual features are presented in Table 5 (available as an additional Excel file).

It is anticipated that Tables 2-4 will provide a framework for the initial assessment of recovery potential and that recovery potential can be assessed on a site and feature-specific basis using Table 5 in addition to Table 4.

Table 2 outlines the underlying ecological and management aims for a particular feature over a specified spatial area. These include consideration of the ecological goals such as long term resilience and sustainability, the impact of management actions to facilitate recovery on other features or other examples of a specific feature and the overall contribution to the national status of the feature. Management considerations include assessment of overall recovery potential, likely recovery timescale and cost, the benefits to society and the impact on stakeholders. Linked to this are the recovery objectives (Table 3), presented for individual species, communities and physical habitats (such as herring spawning ground), acknowledging that recovery of biogenic species incorporates elements of single species and community recovery, as outlined in Section 2.3.

The high level aims and objectives in Tables 2 and 3 are considered in greater detail in Table 4. Table 4 outlines five main considerations listed below, which are not necessarily hierarchical (some are of equal or similar importance), but they can be presented as a stepwise/staged process for determining the cause of decline, recovery potential and the processes which may lead to or inhibit it:

1. the cause of decline and an initial high level assessment of the potential to remove the cause of decline;
2. species and habitat requirements;
3. factors limiting recovery potential;
4. possible measures to improve recovery potential;
5. cost effectiveness.

Generic physical and biological factors influencing recovery are presented as categories in Table 4 and are explained more fully in Section 5. It should also be highlighted that Table 4 may be a useful tool to identify areas where we have inadequate knowledge and that require further research, to improve understanding of the recovery process and potential for recovery for a feature.

Feature specific influential biological and environmental factors are summarised in Table 5, (provided as an Excel file), with detailed evidence being presented in Section 3 for each species, community or habitat. Table 5 also indicates the pressures to which each feature is sensitive, and the way each pressure modifies the environment against the feature's specific habitat requirements. Specific life history traits which may facilitate or inhibit recovery are also presented, together with potential management options.

Table 2. Recovery aims

<b>Aims for recovery of feature</b>	
Underlying ecological considerations	<ul style="list-style-type: none"> <li>• Recovered areas should have the structure and functioning expected of areas with no/minimal local pressures.</li> <li>• They should make a tangible contribution to the national conservation status of the feature</li> <li>• Recovery should not be at the expense of other feature units, i.e. unsustainable translocation.</li> <li>• The baseline/end-point should be stable, sustainable and resilient once intervention is removed</li> <li>• They should contribute to the ecological functioning of adjoining sub-units of the same feature type/improve meta-population dynamics.</li> </ul>
Underlying management considerations	<ul style="list-style-type: none"> <li>• There should be a clear understanding of what the recovered end state should be/look like</li> <li>• Recovery is to be achieved in the shortest possible time</li> <li>• Recovery is to be achieved with the smallest allocation of resources</li> <li>• Recovery should require the manipulation of the minimum number of ecological factors possible</li> <li>• There should be minimal disruption for other marine stakeholders, users and uses</li> <li>• Recovery actions should be undertaken within an ecosystem approach to management</li> <li>• Recovery should be accompanied with an appropriate monitoring programme</li> <li>• There should be a rapid realisation of benefits following recovery/public support for the activity</li> </ul>

Table 3. Recovery objectives

<b>Objectives for recovery of feature based on definitions (Section 2)</b>		
	<b>Recovery/recovering</b>	<b>Recovered</b>
Species	<ul style="list-style-type: none"> <li>• A consistent trajectory, detectable above systemic variability, of net population growth;</li> <li>• Increasing biomass and development of structural population parameters;</li> <li>• Moving towards a range of values, specified by the recovered reference range, for a defined spatial area.</li> </ul>	<ul style="list-style-type: none"> <li>• 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.</li> </ul>
Communities (multiple species)	<ul style="list-style-type: none"> <li>• A consistent trajectory, detectable above systemic variability, of community descriptive parameters towards a range of values, specified by the recovered reference range, for a defined spatial area.</li> </ul>	<ul style="list-style-type: none"> <li>• A stable, enduring similarity, detectable above systemic variability, of community descriptive parameters to the range of values, specified by the recovered reference range, for a defined spatial area.</li> </ul>
Physical habitats	<ul style="list-style-type: none"> <li>• 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 recovered reference range, for a defined spatial area.</li> </ul>	<ul style="list-style-type: none"> <li>• 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 recovered reference range, for a defined spatial area.</li> </ul>

#### **Box 16. Conclusions for assessing recovery potential**

Based on the review in Section 3, individual species and habitat have been placed into one of three recovery categories:

- species/habitats, supported by evidence, with the potential for recovery;
- species/habitats, lacking evidence, but with some potential for recovery; and
- species/habitats, supported by evidence, lacking the potential for a recovery.

Common factors influencing recovery were identified and included:

- ongoing pressures;
- physical condition of the habitat;
- ecological connectivity;
- size, location and condition of the remaining population or habitat before starting the recovery phase;
- intrinsic biological traits such as fecundity, growth, dispersal and longevity; and
- interactions with other species.

A framework has also been provided for the assessment of the cause of decline and recovery potential.

Table 4. Stages of assessment of the cause of decline and recovery potential. References to relevant sections of the report are provided for additional information.

Considerations for determining recovery potential	Relevant section	Notes
<b>STAGE 1- Assessing cause of decline and recovery potential</b>		This first stage aims to determine, on a site-specific basis, the cause(s) of the decline of a species or habitat, the disturbance history, and whether management of the causes of decline is possible. Once these components, as outlined in Stages 1a-d, have been assessed, they can be used to provide an initial indication of whether the decline can be halted and if the features are in a state from which recovery may be possible, in combination with further assessment of the factors in <b>stages 2 and 3</b> . Stage 1 can also assist with the prioritisation of sites under consideration for management action, in particular by identifying sites where management would not successfully overcome natural causes of decline. Decisions could be made at this stage about whether or not to continue with further assessment (stages 2-5).
<b>STAGE 1a – What is the cause of concern?</b>		
Is adult/juvenile mortality occurring? (species/biogenic habitats)	3	<b>The cause for concern</b> is identified based on site-specific evidence of decline in terms of the attributes outlined under stage 1a. The causes for concern should be listed for the site and evidence sources documented.
Is reproductive activity poor? (species/biogenic habitats)	3	
Is recruitment occurring or not? (species/biogenic habitats)	3	
Has the spatial extent of populations, communities and biogenic habitats been reduced?		
Have the physical attributes of the habitat changed?		
Has the spatial extent of the physical habitat reduced?	3	
<b>STAGE 1b – What are the cause(s) of decline?</b>		
What can the decline be linked to? E.g. anthropogenic pressure/activity (near-field or far-field), natural population variability, climate change, change in habitat use/spatial distribution, disease, parasitism, competition/interaction with other species?	3, 5	<b>The cause of decline</b> is assessed by outlining any evidence which exists of relevance to the site, for links between the cause of concern (e.g. increased adult mortality) and anthropogenic activity/pressures, climate change, and biological mechanisms of decline e.g. natural population variability, natural change in habitat use, systematic and stochastic changes in an area used by a species or community, disease and parasitism and

Considerations for determining recovery potential	Relevant section	Notes
		wider scale population decline.
<b>STAGE 1c – Assess the disturbance history and degree of decline</b>		
What is the nature of the disturbance? For example, is it patchy, with remaining intact patches of habitat/species and some degree of habitat heterogeneity, or is the disturbance widespread and/or associated with total removal of the habitat with extensive habitat homogenisation?	5.1.1	<p><b>The disturbance history</b> should be documented as far as known. This identifies the ecological starting point for recovery and helps to assess the potential for recovery. A description should be given using the bullets provided under Stage 1c as guidance. The disturbance history strongly influences the potential for recovery, for example areas where widespread species removal and habitat homogenisation has taken place are less likely to recover (or the recovery process will be significantly protracted) compared to areas where the disturbance is patchy, areas of remnant organisms and suitable habitat remain and a degree of habitat heterogeneity remains.</p>
Are there concerns regarding frequency/intensity/spatial extent of disturbance?	5.1.1	
<b>STAGE 1d – What is the potential for removing the cause of decline?</b>		
Is the decline due to a manageable or unmanageable natural/anthropogenic pressure?		<p>This is a high level assessment of the <b>potential to remove the cause of decline</b>. For the identified anthropogenic or natural pressures identified in stage 1b, this stage assesses whether these (i.e. the cause of decline) are potentially manageable or unmanageable. If the cause of the decline is of anthropogenic origin and is manageable there is more likelihood of removing the cause of the decline and for recovery occurring.</p>
<b>STAGE 2 Review the feature's habitat requirements</b>		
What are the habitat requirements of the feature?		<p>The specific habitat requirements of the individual features are presented in Lancaster <i>et al.</i> (2014), Section 3 of this report and in Table 5, alongside the changes to habitat quality associated with specific pressures. This stage should assess what the feature requires for growth and survival and what the conditions are at the site.</p>

Considerations for determining recovery potential	Relevant section	Notes
<b>STAGE 3 Determine the factors limiting recovery potential</b> <i>Refer to Table 5 for feature-specific information</i> <i>(note that not all the factors below are relevant for all features)</i>		
<b>Environmental</b>		
<i>Habitat attributes</i> <ul style="list-style-type: none"> <li>• Habitat type - has the habitat completely changed, is it no longer appropriate and if so, what is the potential for physical recovery? E.g. has homogenisation of mixed sediments occurred, resulting in inappropriate structural sediment conditions and what is the potential and likely timescale for physical recovery?</li> <li>• Habitat extent and availability – is there sufficient area of suitable habitat to support recovery of the feature to enable specific recovery objectives relating to spatial extent?</li> <li>• Depth</li> <li>• Temperature</li> <li>• Salinity</li> </ul>	5.1.2	<p>Stage 2 identified the habitat requirements of the feature and the site conditions. Stage 3, assesses which specific factors are limiting the recovery of the feature at the site, not only with regard to the environmental attributes but also biological factors.</p> <p>Firstly the environmental parameters of the site should be compared to the features' requirements with regard to habitat attributes, sediment, hydrodynamics, water quality and residual pressures, as detailed. This will help determine whether the physico-chemical structure of the habitat can provide suitable conditions for the re-establishment of a particular species, community or physical habitat structure. This should initially include simple measures of depth, hydrodynamic conditions and sediment structure (e.g. particle size, organic content) but for some species (e.g. seagrasses and species inhabiting stable muds), the degree of consolidation and chemical gradients within the sediment may also be important. These factors are directly linked to the causes of decline and the disturbance history assessed in stage 1.</p> <p>Secondly, the biological components of the habitat upon which the recovery process may be dependent include the initial status of the population, life history traits, biological connectivity and biological interactions e.g.</p>
<i>Sediment - physical</i> <ul style="list-style-type: none"> <li>• Textural sediment class; particle size distribution; sediment structure, sorting; cracks, crevices and interstitial spaces; consolidation and stability; rugosity; and organic content</li> </ul>	5.1.2	
<i>Sediment - chemical</i> <ul style="list-style-type: none"> <li>• Redox potential / depth of RPD; establishment of chemical gradients</li> </ul>		
<i>Sediment – biological</i> <ul style="list-style-type: none"> <li>• Establishment of microbial communities, and meiofaunal and macrofaunal communities upon which the recovering feature may be dependent.</li> <li>• Establishment of predator-prey relationships at trophic levels below that of the recovering species.</li> </ul>		

Considerations for determining recovery potential	Relevant section	Notes
		<p>predator-prey interactions at trophic levels below that of the feature in question. Information regarding the feature, its biological traits and any site specific information relating to the factors outlined should be detailed.</p> <p>Further detail, specific to each species or feature, of these factors is provided in <b>Table 5</b> and the subsections of <b>Section 5</b>.</p>
<p><i>Hydrodynamic attributes</i></p> <ul style="list-style-type: none"> <li>• Water movement and residence time – are MPAs, or features in areas with little water exchange and therefore low potential for propagule dispersal/receipt? Is the presence of physical structures or seabed topography influencing the microscale hydrodynamic processes that may be required to maintain the required habitat conditions?</li> </ul>	5.1.4	As above
<p><i>Physical/hydrodynamic connectivity</i></p> <ul style="list-style-type: none"> <li>• Do the prevailing current patterns allow connectivity between species and habitats?</li> <li>• What is the distance between populations/suitable settlement sites in relation to larval survival time in the water column?</li> <li>• Is habitat homogenisation such that primary and secondary settlement sites no longer exist or are too far apart?</li> <li>• Are there physical barriers to dispersal and migration?</li> </ul>	5.1.3	
<p><i>Water quality</i></p> <ul style="list-style-type: none"> <li>• Dissolved oxygen; salinity; nutrients; suspended solids/light penetration; phytoplankton and zooplankton; and chemical contaminants</li> </ul>		
<p><i>Appropriate management area</i></p> <ul style="list-style-type: none"> <li>• Does it account for near and far field influences and are the management measures appropriate?</li> </ul>	5.1.2	
<p><i>Residual pressures following management action</i></p> <ul style="list-style-type: none"> <li>• Residual chemical contaminants and organic enrichment</li> </ul>	5.1.1	

<b>Biological</b>		
<i>Initial status of population and status of surrounding, influential populations (brood stock)</i> <ul style="list-style-type: none"> <li>• Is there sufficient density of reproductively active individuals (or is there evidence of recruitment) within a metapopulation to provide sufficient larval exchange and subsequent population growth? Is there a brood stock?</li> </ul>	5.1.1; 5.1.5	As above
<i>Biological connectivity</i> <ul style="list-style-type: none"> <li>• evidence for larval exchange between populations;</li> <li>• distance between populations in relation to larval survival time.</li> </ul>	5.1.3; 5.1.4	
<i>Life history traits</i> <ul style="list-style-type: none"> <li>• growth; age at sexual maturity; longevity; spawning strategy and frequency; fertilisation success; juvenile mortality; larval dispersal ability; mode and range; and adult mobility</li> </ul>	5.1.6	
<i>Biological interactions</i> <ul style="list-style-type: none"> <li>• predation; competition; disease; parasitism; bioturbation, biostabilisation, bioengineering; ammensalism; commensalism; and mutualism.</li> </ul>	5.1.7	
<i>Poor knowledge of feature biology</i>	3	
<i>Change in the spatial distribution of the feature</i>		
<i>Wider trends in population status</i>		

Considerations for determining recovery potential	Relevant section	Notes
<b>STAGE 4 Measures to address the cause of the decline and improve recovery potential</b>		
<b>Environmental</b>		
Pressure removal - near and far-field pressures	3	Where evidence of barriers to recovery can be found, active intervention may be appropriate to remove these. It must be acknowledged though that many features, particularly those considered in this study, are long lived and that the recovery process will take time, even under optimum conditions.
Change to the management area/scale	5.3	
Change to the timing of management measures	5.3	
Changes to the activity	5.3	
<b>Biological</b>		
Habitat enhancement	5.3.1	At this stage consideration should be given to the potential for additional management of the physical and biological processes, together with active intervention to remove pressures and/or restoration techniques. A brief review of the potential value of these techniques is given in Section 5.3, with specific information for features given in Table 5 and Section 3.
Brood stock protection	5.3	
Hatchery cultivation	5.3.3	
Translocation	5.3.2	
Disease / Predator control	3; 5.3	
<b>STAGE 5 Assess cost effectiveness of measures</b>		
Decisions over the cost-effectiveness of implementing measures to facilitate recovery is not an area of SNH expertise and Marine Scotland would normally take the lead in making assessments regarding the costs versus benefits of management relating to MPAs.		

## 5. SUMMARY OF FACTORS INFLUENCING RECOVERY

### 5.1 Ecological factors

This section summarises the common external, environmental biological factors which have been found to influence recovery in the species and habitats described in Section 3, and draws upon the wider evidence from other species and habitats. This ensures inclusion of relevant biological influences that may not have been documented in the studies reviewed in Section 3.

#### 5.1.1 Scale of disturbance and degree of impact

The scale of the disturbance and the state of the community following disturbance (i.e. degree of damage) are important factors in determining recovery potential and rate, particularly for biogenic species (or other species that modify the habitat), colonial organisms and species exhibiting density-dependent growth (e.g., saltmarsh and seagrass) (Angelini and Silliman, 2012). In a review of recovery in intertidal environments, Mazik and Smyth (2013) found that whilst recovery rates were highly variable, there was a general trend of increasing recovery time with increasing spatial scale and intensity of the disturbance. Angelini and Silliman (2012) suggested that the non-linearity between the perimeter of the disturbed patch and the internal area would result in slower rates of colonisation in large areas because of slower rates of lateral colonisation by adults or propagules, and lower levels of protection from predation by the surrounding community and other physical stressors. Therefore recovery times following large scale disturbance may be protracted or full recovery may not even be achievable. Small-scale experimental studies in intertidal soft sediments have indicated recovery times of a few months compared to a few years after large-scale anthropogenic disturbance (Mazik and Smyth, 2013). Indeed, Trigg and Moore (2009) found evidence of recovery of *Limaria hians* following small scale defaunation but predicted that recovery following large-scale defaunation, associated with scallop dredging, would be prolonged and may take over a century. This also applies to *Zostera* species where small scale patchy disturbance may be followed by recovery over a few years but recovery may not be possible following large-scale disturbance or complete removal of a bed (Neckles *et al.*, 2005; Boese *et al.*, 2009).

Where there is only partial removal of the species or habitats of interest, the recovery process may be faster as the remaining organisms provide an immediate source of colonists, assuming that they are undamaged and that the physical conditions of the habitat are suitable for their survival. This applies to *Limaria hians*, where lateral growth rates of the *L. hians* bed were estimated to be much longer for areas that had been extensively disturbed compared to those associated with small scale disturbance (Trigg and Moore, 2009). In subtidal rocky habitats, Bevilacqua *et al.* (2006) state that where disturbed areas are characterised by fragments and small colonies of remaining organisms (a species legacy), lateral growth from the remaining colonies are likely to be the main sources of recolonization followed by larval supply from colonies outside the restoration area.

The presence of these remnant organisms may increase the opportunity for recolonisation (for some species). However, the resilience of these remnant patches is size dependent, particularly for reef forming or colonial organisms, with small patches having a comparatively high proportion of exposure to physical stress around the edges (Angelini and Silliman, 2012). Roberts *et al.* (2003) emphasise that a higher degree of persistence and resilience is generally associated with well replicated and connected habitat or sub-population units, and may not always simply be a reflection of size.

### 5.1.2 Physical recovery of the habitat

With the exception of black guillemot, the most significant pressure, posing the greatest risk to features of conservation interest in Scottish coastal waters is that of physical damage to, or removal of habitats. This pressure is predominantly, but not exclusively, associated with fishing activities (predominantly those employing mobile, demersal gear such as trawls, dredges and hydraulic methods), renewable energy developments and, to a lesser extent, infrastructure development (Scottish Government, 2013). Physical disturbance not only results in removal of or damage to sediment dwelling organisms, it also modifies the physical structure of the sediment through changes in grain size distribution and sorting, organic content, microbial content, compaction, consolidation and overall mobility and chemical gradients/redox profiles. The close relationships between the physical properties of the substratum and the community type inhabiting it are well documented and are reviewed in detail in Snelgrove and Butman (1994) and Gray and Elliott (2009). Therefore, understanding the timescale of physical, habitat recovery is of relevance to understanding the timescale for biological recovery. For species that also constitute the physical substratum (e.g. maerl) biological factors such as growth rate and the degree of fragmentation dictate the time scale for recovery.

Broadly, the physical structure of the sediments is governed by hydrodynamics (in the absence of any other pressure). However, at more localised levels, biological activity plays an important role. For example, at a local level the biological processes of bioturbation and colonisation by particular functional groups of species will influence the way in which the sediment becomes favourable for colonisation by other species. Schratzberger and Warwick (1999) stated that hydrodynamic conditions were most important in unstable sands, whereas biological interaction was likely to be more important in sheltered muddy environments. Thrush *et al.* (1996) attributed prolonged macrofaunal recovery times (i.e. greater than the 9 month observation period for the study) to sediment instability following the removal of tubicolous polychaetes, which stabilise sediments and therefore have an important role in maintaining habitat structure.

Prolonged physical recovery times have been documented in maerl habitats following scallop dredging, with dredge tracks still being visible 5 years after dredging has ceased (Hall-Spencer and Moore, 2000b). It was emphasised that recovery of the physical structure of maerl beds was not indicative of biological recovery (of maerl) and recovery of the associated benthic community was not studied. Furthermore, because of the sensitivity of maerl, the scale of the experiment was limited to a single tow of three dredges compared to commercial boats which typically tow 14-16 dredges. Therefore, the extent of the impact is likely to be far greater in areas of commercial dredging and the subsequent physical recovery times very prolonged (hundreds of years)

Foden *et al.* (2009) estimated the rate of biological recovery, following aggregate extraction, for assemblages on coarse sediments (including gravel substrata) in areas differing tidal stress. Achievement of a recovered physical state within weak, moderate and strong tidal energy sites was, based on a thorough review of the literature, estimated to be approximately 12 years, 20 years and 5 years respectively. Mean biological recovery times ranged from 5 - 12 years, although variability was high and the definition of biological recovery and the definition of the recovered reference range for the habitats reviewed was not clear. Whilst gravel extraction activity is not widespread in Scottish waters, there are similarities between the effects of this, and the use of mobile demersal fishing gear, on the physical structure of the sea bed. Therefore, these recovery times provide a general indication of the recovery timescales which can be expected for coarse sediments exposed to a removal pressure.

### 5.1.3 *Habitat fragmentation, homogenisation and connectivity in relation to the hydrodynamic regime*

Human activities and their associated pressures on the environment can lead to fragmentation of habitats and species, thereby reducing the size and connectivity of habitats (Ewers *et al.*, 2006) and increasing the isolation of (generally small) habitat patches in which ecological processes differ to those in larger surrounding, continuous habitats (Hovel and Lipcius, 2001). Ultimately, this leads to shifts in biodiversity associated with the loss of habitat-specific or functionally important species (Thrush *et al.*, 2008). Fragmentation leads to reduced connectivity between local and regional species pools, with a restricted supply of recruits due to the distance between patches, or the presence of environmental barriers preventing dispersal (Thrush *et al.*, 2008). Connectivity refers to the natural linkage between marine habitats which occurs via larval dispersal and the movement of adults and juveniles (Roberts *et al.*, 2003). Thrush *et al.* (2008) described two types of connectivity:

1. **environmental connectivity** and the influence of physical processes on recovery, where hydrodynamics, physical barriers and the rate of immigration and emigration need to be considered. Large-scale hydrodynamic conditions, together with local benthic boundary layer conditions influence the erosion and deposition dynamics, the broad-scale supply of colonists and the ability of those colonists to settle.
2. **ecological connectivity** is measured as  $\beta$  diversity which is the difference between regional or habitat ( $\alpha$ ) diversity and overall site species richness.  $\beta$  diversity is considered an important measure linking local and regional diversity with low dissimilarity between sites indicating high connectivity.

Hydrodynamic conditions (environmental connectivity), to a large extent, control connectivity between sub-populations and habitats and will strongly influence the likelihood of settlement and recruitment from surrounding sub-populations. Similarly, the presence of physical barriers to larval transport and adult migration (which may be natural or anthropogenic) should be considered when determining MPA management areas, together with any necessary management measures. Furthermore, the degree of habitat damage should be assessed with those areas with the highest potential for physical recovery being given priority in terms of protection.

Because of the spawning and dispersal mechanisms of many marine species, it is generally assumed that broad-scale dispersal potential and connectivity is high. Thrush *et al.* (2008), challenged this theory, indicating that the dispersal ability of most benthic species is limited and is not simply influenced by wide-scale hydrodynamic processes or distance to nearest neighbours (i.e. similar or target community type). Additionally, Gallego *et al.* (2013) emphasised the importance of larval behaviour (e.g. duration of the pelagic phase) and spawning time, highlighting the potential for transport to unfavourable habitats and larval mortality before settlement. Zajac and Whitlatch (2003) found the potential for organism transport to be greater in sandy habitats than muddy or gravelly habitats, whilst Lundquist *et al.* (2006) found organism transport and deposition to be greatest during periods of wind-induced wave activity.

Larval settlement at a site is the initial stage of recruitment to an adult population. Documenting the supplies of larvae to a site is an important first stage in securing recovery. Based on the value of passive restoration and the adequate supply of propagules that underpin this strategy, management efforts need to consider both the target area for restoration and the surrounding areas that contribute to that area. Certain neighbouring populations may be disproportionately important due to either their size or hydrodynamic connectivity. These neighbouring populations can be particularly important as a source of propagules to the recovering area. As such, it is also important that these 'source' populations of propagules are also identified and protected.

Hydrodynamic and ecological connectivity are particularly limiting factors for species (i) of low mobility, (ii) those with low fecundity and (iii) those which do not have a pelagic dispersal phase. This is particularly applicable to maerl (Hall-Spencer and Moore, 2000b) and seagrasses (Boese *et al.*, 2009). Additionally, species brood their young (e.g. some holothurians) are assumed to have low fecundity, whilst others are broadcast spawners (O'Connor, 1981). Lack of connectivity has also been demonstrated between populations of common skate as a result of generally low adult mobility, temperature regime preference and low/infrequent fecundity (Little, 1998; Griffiths *et al.*, 2010a). The recovery potential of black guillemots is also suggested to be reduced by poor dispersal and the loss of colonies. The loss of colonies decreases the level of connectivity between breeding areas and ultimately isolates populations that might have otherwise provided a supply of adults for the enhancement of depleted areas.

Thrush *et al.* (2008) found habitat recovery rates to be controlled by landscape features linked to fragmentation. This study emphasised the importance of interaction between communities at local and wider scales. Recovery at patch scale is linked by dispersal potential, the nature of the regional species pool and the environmental setting of the disturbance. Ecological recovery rates were found to decrease with increasing fragmentation with the fastest recovery rates being associated with the smallest ecological nearest neighbour distance (i.e. those closest to similar communities) and the lowest number of physical barriers to dispersal of colonists, coupled with biological factors such as reproductive strategy.

#### 5.1.4 *Spatial extent, distribution and condition of species and habitats*

Assessment of the spatial extent of the meta-population and the spatial extent and distribution of the component populations, within the dispersal range of the species in question, is necessary as an indication of the potential sources of, quality and abundance of propagules. Populations that supply propagules to other populations are referred to as source populations. Recipient populations, unable to provide enough localised recruitment and therefore sustain itself over time, are referred to as sink populations. It is important to determine whether or not the surrounding populations are reproductively active and net exporters of larvae, i.e. source populations that might sustain sink populations elsewhere or whether they act as both sources and sinks. The state of the existing feature should be considered in terms of its actual or potential suitability for recruitment.

If connectivity remains between sub-populations, then an overall analysis of adult population size will benefit the predictions of recovery. The population structure also has important implications for overall fecundity. If the population structure is highly variable between year classes, it would suggest that recruitment between years is sporadic and maybe dominated by mast years (years of extremely high reproductive output). If the population is dominated by older age classes then this may indicate a shortage of recruitment of new recruits. Age skews may also have important implications for overall fecundity if older individuals are associated with differing levels of gamete production.

#### 5.1.5 *Life history, dispersal ability and recruitment*

##### 5.1.5.1 *Fecundity and fertilisation*

The life histories of many of the species under consideration in this document are such that recovery potential may be limited by biological factors, even in the absence of additional external factors. In bivalves, spawning is variable between and within species (depending on local environmental conditions) and can occur as a single event, a protracted spawning period or as several spawning periods (Gosling, 2003). Bivalves tend to release large numbers of eggs into the water column (Gosling, 2003) which develop into lecithotrophic (yolk dependent) larvae, giving bivalves the advantage of wide dispersal of potentially large

numbers of larvae. This enhances the potential for recruitment, rapid population growth and therefore, recovery from disturbance. However, despite the high fecundity in bivalves, fertilisation rates (and thus recruitment) can be low (Seed and Brown, 1975). This is particularly relevant to *Modiolus modiolus* where low fertilisation success has been documented in relation to hydrodynamic conditions (Eläßer *et al.*, 2013), density dependent factors (Kennedy and Roberts, 1999) and spawning frequency (Wiborg (1946 in Brown, 1984)). For some bivalves of relevance to this study, fertilisation success is not known (e.g. *Atrina fragilis*, *Arctica islandica*).

The timing and synchrony of reproductive bouts are important for overall reproductive success. Populations. For example, *M. modiolus* is a trickle spawning species. If there are not enough adults spawning simultaneously, gamete concentration may be too low for high encounter rates and fertilisation success. Larval duration in the planktonic phase is another important factor in determining dispersal and the potential for 'source and sink' connectivity between populations of the same species. Species with a short planktonic phase are likely to have a reduced dispersal potential and may contribute less to propagule supply for other populations. Conversely, long planktonic development phase e.g. 30-50 days in the case of *Arctica islandica*, has the potential advantage of wide dispersal, but it carries a risk of transportation away from favourable habitat.

For some species, synchronising the release of gametes is an effective method of concentrating gametes and thereby overcoming dilution in the water column and resulting poor fertilisation rates. Within sheltered sea loch environments, *M. modiolus* shows less pronounced reproductive bouts and functionally becomes a trickle spawner (Seed and Brown 1975). At larger population levels within enclosed water bodies this is unlikely to be a disadvantageous strategy. However, as the brood-stock is heavily depleted, this leads to gametes occurring at a much lower concentration, leading to greater dilution of gametes and consequently lower fertilisation rates.

Information relating to the reproductive biology of echinoderms is limited and very few details relating to *Leptometra celtica* (Crionidae: Antedonidae) or *Neopentadactyla mixta* were found. However, Chadwick (1907, in Nichols, 1991) suggested that *Antedon bifida* (Antedonidae) retention of embryos by the brooding females, which hatch into free-swimming larvae after 4 or 5 days. No information on fecundity was found although the brooding behaviour implies low fecundity compared to bivalves. It is emphasised that the degree of similarity between the reproductive strategies of *A. bifida* and *L. celtica* is not known.

Extremely low or no fecundity has been reported for maerl (Cabioc, 1970) and given that reproduction is primarily through fragmentation, the dispersal ability of both species of maerl is limited.

The sandeel *A. tobianus* produces between 4000 and 20,000 eggs which develop in the sediment before hatching (Rowley and Wilding, 2008), highlighting the importance of substratum type to this species. Therefore, assuming successful recruitment, the recovery potential of sandeels is considerable. In contrast, low fecundity and low dispersal potential are characteristic of the longnose skate and this may present an important barrier to the recovery this and other skate species (Griffiths *et al.*, 2010).

Thrush *et al.* (2008) emphasised that, regardless of fecundity and potential dispersal ability, the role of dispersal in recovery is mediated by other environmental and ecological factors operating at a range of spatial scales, including habitat selectivity, growth rates and species interactions.

#### 5.1.5.2 Dispersal ability and recruitment

For species with a pelagic larval phase, that rely on passive dispersal mechanisms, hydrodynamic conditions and the time it takes them to reach metamorphosis, will control the fate of larvae by influencing the distance and direction of transport. Ultimately, these factors dictate whether or not larvae are transported to suitable habitat. Recruitment following spawning and settlement is dependent upon the ability of larvae to select appropriate substrata (Santos and Simon, 1980). According to Woodin *et al.* (1995), post-larval juveniles of three infaunal species (including the bivalve *Mercenaria mercenaria*) were able to discriminate between disturbed and undisturbed sediments and showed a reluctance to burrow in physically disturbed sediment, resulting in re-entrainment of the larvae. Whilst the potential for settlement and recruitment elsewhere was increased by this strategy, it also implies that the recovery potential of disturbed sediments may be compromised. Beukema *et al.* (1999) also suggested that favourable sediment conditions were likely to strongly influence colonisation by adults which (in benthic species) are generally reliant on active dispersal (crawling or burrowing) compared to the passive dispersal strategy of the larvae.

In the case of *Modiolus modiolus* and *Ostrea edulis*, it has been demonstrated that larval recruitment is dependent on the presence of adults, which provide a favourable substratum for settlement (Roberts *et al.*, 1975; Mann and Powell, 2007; Rees *et al.*, 2008) and that settlement amongst conspecifics may enhance juvenile survival through protection from predation. In an area where significant reef removal (with respect to these two species) has occurred, poor larval settlement may contribute to poor recovery.

Finally, larval and juvenile mortality can be high (Newell *et al.*, 1998; Witbaard and Bergmann, 2003), potentially limiting settlement and recruitment in areas that are poorly connected (e.g. large distance between suitable habitat areas).

#### 5.1.5.3 Slow growth and age at sexual maturity

The majority of the species with a recover or conserve objective are relatively (some extremely) long-lived and slow growing, reaching sexual maturity only after several years. This may be a strategy to increase survival whereby energy during the early years of life is diverted towards rapid somatic growth enabling the organism to become sufficiently large to be inaccessible to predators (e.g. *Arctica islandica*, Ridgway *et al.*, 2012). Whilst this strategy has benefits for long term survival, it means that there may be a significant time delay between settlement and spawning so that recovery times (in terms of a self-sustaining, functional population or community) will be protracted. Details of longevity and age at sexual maturity for priority features with regard to this project are summarised in Annex 2. With respect to bivalves, a number of species reach sexual maturity within their first year of life (e.g. *Ensis* spp., *Timoclea ovata*, *Chamellea gallina* and *Spisula solida*), indicating that settlement, recruitment and recovery of a self-sustaining population (relying on interaction between sub-populations) of these species may be achievable within a few years, assuming a sufficient degree of physical recovery of the substratum to enable recruitment. In contrast, *A. islandica* is extremely long-lived and can take up to 14 years to reach sexual maturity (Thompson *et al.*, 1980). Biogenic reef forming species, in particular *Modiolus modiolus*, take several years to reach sexual maturity implying that recovery time would be protracted. Whilst there is little documented recovery of these species, Trigg and Moore (2009) demonstrated that *L. hians* had the potential to recover but that the recovery process may take >100 years where large-scale physical disturbance had taken place. Faster recovery (<10 years) times were documented by Minchin (1995) following chemical contamination although in this case, there was no physical alteration of the sediment. It is of note that this species is thought to reach sexual maturity within 2 years (Minchin, 1995).

#### 5.1.5.4 Adult mobility

Whilst the larvae of many of the species reviewed here have high dispersal potential, the adults do not. For example, seagrasses, the cnidarian *Caryophyllia smithii* and maerl are particularly immobile and the dispersal range for adult *Ostrea edulis*, *Attrina fragilis* and *Modiolus modiolus* follows the order 0 m, 1 m and 10 m (MarLIN, 2006). Other bivalves have a greater degree of mobility during adulthood and Hall-Spencer and Moore (2000a) suggested that adult *L. hians* were able to swim, indicating some potential for dispersal and recolonisation for disturbed or displaced individuals, assuming suitable habitat can be found. However, the distance over which this species is able to swim is not known and is likely to be limited by the size of the organism and the energy required for propulsion (i.e. clapping its shells together to create a jet of water, in combination with a rowing motion of its tentacles (Tyler-Walters, 2008). MarLIN (2006) suggested that the dispersal range of adult *A. islandica* was 100-1000 m and it is unlikely that the adult dispersal range of *L. hians* is any greater than this. The echinoderm *Neopentadactyla mixta* is reported to have an adult dispersal range of 10-100m (MarLIN, 2006). No information was available on the dispersal ability of *Funiculina quadrangularis*, *Swiftia pallida* although for other seafans (Octocorallia) such as *Eunicella verrucosa* (pink seafan), adults have been reported to have no dispersal ability (MarLIN, 2006). Information on the mobility of *Leptometra celtica* was not found although observations by SNH (pers. comm.) indicate that adults can swim but the distance is unknown.

Adult fish can be highly mobile but Little (1998) described the common skate as a relatively sedentary species, observing horizontal movement to be limited to 20 km. Additionally, herring have been found to re-visit their spawning grounds (Smith and Morse, 1993) so that, whilst there is potential for interaction between populations, their ability to use alternative spawning grounds is unclear. Therefore, in the context of recovery potential, these species are considered to have low mobility. Similarly, black guillemot are described as relatively sedentary birds with movement away from their colonies being limited to an area of around 10.5 km (Okill, 2002).

It is evident that recovery of all of the species considered here will be limited by poor adult mobility in terms of inability to escape disturbance (generally for benthic species) and inability to colonise new habitat.

#### 5.1.6 Biological interactions

At all stages of life, marine species are susceptible to predation, disease and competitive interactions. In fully or partially defaunated habitats, recolonisation is strongly influenced by the impact that newly colonising or relict organisms have on the sediment structure and on each other. Bioturbation by one species can lead to resuspension or smothering of another, potentially altering its exposure to currents, wave action and predation (Zajac, 2004). Similarly, the presence of bioturbating species such as *Arenicola* has been shown to limit the potential for *Zostera* recovery (Valdemarsen *et al.*, 2011). Modifications to the sediment (stabilisation, destabilisation, particle size and organic matter distribution and oxygenation) by one species may make it more or less favourable to another (Volkenborn *et al.*, 2009) such that the order of colonisation could influence the overall community structure in time. Ammensalistic and commensalistic interactions (in rocky and soft sediment habitats) also influence the way a community develops and its recovery potential whereby a particular species may be key to the existence of another and vice versa (Beukema *et al.*, 1999) and, through a process known as 'gardening', the production of faecal material can enhance the microbial content of the sediment making it favourable for other species (Hylleberg, 1975).

#### 5.1.6.1 Consideration of systematic and stochastic changes in an areas use by a species or community

The environmental conditions within an MPA may not remain consistent through time and consequently MPAs may become sub-optimal. For example the Plaice Box, an area approximately 40,000 square kilometres large, situated north of the Dutch and German Wadden Islands and west of the Danish Wadden Islands, was established in 1989 to protect undersized plaice (van Keeken *et al.*, 2007; Beare *et al.*, 2013). Originally closed to large beam trawlers during key periods of the year, in 1994 this prohibition was extended to cover the whole year. However, the Plaice Box has had limited effectiveness because, since its inception, juvenile plaice have moved further offshore, outside of the protected area. It has been suggested that this movement has been the result of increased water temperatures (acting either directly by exceeding the tolerance range of the juvenile fish, or indirectly by increasing the plaice food requirements above that supported by the available food resources; van Keeken *et al.*, 2007).

#### 5.1.6.2 Timescale of recovery in relation to recovery objectives

In intertidal and subtidal soft sediment habitats (including gravel), documented recovery times for various species and communities range from a few months (Hall and Harding, 1997) to over 25 years (González-Correa *et al.*, 2008) with predicted recovery times for some exceeding 100 years (Trigg and Moore, 2009). Lotze *et al.* (2011) highlighted cases where seagrass beds showed little recovery 100 years after disturbance (trawling) ceased. However, as highlighted in section 2, the time taken to achieve 'recovered' status is strongly influenced by the parameters used to determine recovery and the degree of damage (e.g. partial vs. complete defaunation).

It is generally accepted that recovery in coarse sediments occurs more quickly than in fine sediments (Mazik and Smyth, 2013). For black guillemot (*Cepphus grille*), recovery times of between 1 and 5 years have been reported (Heubeck, 2000) whilst recovery times for skates (*Dipturus* spp.) is reported to be prolonged (many years) as a result of high sensitivity to fishing and aspects of the life history of this genus (Neal *et al.*, 2008). Lotze *et al.* (2011) indicated that slow growing sponges and corals may recover after 8 years (although the geographical context is not known) and that clupeids (e.g. herring) may recover within 5-10 years, depending on the circumstances of their depletion.

Whilst these recovery times are variable according to specific habitat type, communities where recovery has been documented are largely composed of small and relatively short-lived species with high fecundity. In contrast, the species under consideration in this study are generally long-lived with low fecundity, often poor and infrequent recruitment and, for many species, poor dispersal ability. Therefore, recovery times (if indeed recovery is possible) will be protracted. For biogenic habitats, recovery has not extensively been documented. Based on species longevity, expert judgement suggests that recovery may take over 100 years, assuming that observed recovery rates continue in a linear fashion (i.e. the accumulation of associated species required for the recovered reference range increases linearly over time rather than requiring progressively more time for the re-establishment of the rarer species). For other species (particularly fish), population decline and recovery are difficult to ascertain against a background of high variability (natural or otherwise). It is important that monitoring programmes and conservation goals take this into consideration, together with natural variation in population density and community structure. In particular, variability in the reference conditions, fluctuations during the recovery process and the influence of changing natural and anthropogenic pressures must be considered, together with the potential for changes to the legislation, management options and conservation priorities.

## 5.2 Management considerations

In a review of evidence for recovery, Lotze *et al.* (2011) indicated that in 95% of cases, recovery had occurred directly as a result of the reduction or removal of pressures (associated with human activities) that led to the degradation of habitats or depletion of species. This approach 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 these 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. Whilst management measures are an important driver of recovery, the physical and biological processes highlighted in section 5.1 need to be considered for management to be effective.

Most of the species and habitats reviewed here are long lived with low fecundity and / or limited recruitment success, low adult mobility and specific habitat requirements. Due to the small number of cases where any degree of recovery has been documented, it is difficult to identify circumstances that may lead to maximum recovery potential. It is also likely that factors such as MPA size, the location of boundaries, the scale, and nature of management responses will be site-specific. It is, however, apparent that, physical pressures associated with fishing activity pose the greatest threat to most species and habitats considered. For the relatively sedentary benthic species, complete removal of these pressures at appropriate spatial scales, over long time periods is likely to be necessary to achieve any degree of recovery. Despite these issues, and the limitations of the literature associated with MPA design, it is suggested that the following factors are considered, in the order in which they are presented, when determining management for recovery within MPAs.

### 5.2.1 Activities and pressures

An overview of the way management will be dealt with in Nature Conservation MPAs is provided in the draft management handbook<sup>3</sup> produced by Marine Scotland. The handbook outlines that there is an assumption of sustainable use of a site by multiple interests. However, activities which are not compatible with the conservation objectives features within a NC MPA will be managed to ensure that the conservation objectives can be achieved.

A risk-based approach to achieving the conservation objectives of features within a site is being taken when developing management. This involves site-by-site consideration of the protected features, the conservation objectives, the activities that take place in, or in close proximity to, the NC MPA and the sensitivity of these features to pressures associated with these activities (as described in FEAST, Scottish Government, 2013). Where there are overlaps between protected features and activities associated with pressures to which the feature is sensitive, management options have been developed. These options are: management to remove or avoid pressures, reduce or limit pressures or suggest no additional management is required. Any management would be applied on the protected feature under consideration. For example where there is a feature that is sensitive to pressures associated with demersal towed/active gear, with a 'recover' conservation objective, SNH would be recommending that these activities and their associated pressures were removed/avoided in the areas containing this feature, in order to prevent further damage and decline of the feature's condition, and to promote the feature's recovery. The recommendation for the same feature in another site with a 'conserve' conservation

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<sup>3</sup> <http://www.scotland.gov.uk/Resource/0041/00417218.doc>

objective may be the same, in order to ensure the conservation objective is achieved in the long term.

This information has been presented in Management options papers<sup>4</sup> developed for each site by SNH. These will be refined through discussions with stakeholders as required. They also will provide the starting point for the development of a management plan, or a Marine Management Scheme, which will be used to set out the conservation objectives and proposed management for individual NC MPAs.

#### 5.2.1.1 Direct (near-field) pressures

There is a significant lack of evidence of recovery for most of the species that have a conserve or recover objective in Scottish waters. This can, in part, be attributed to the fact that recovery simply has not been widely studied. However, the life histories of most of the species under consideration in this study are such that recovery potential is extremely limited and due to the long time scales required to achieved a recovered endpoint it may take time from the implementation of management measures for their to be evidence of recovery. Where evidence for recovery has been documented (e.g. *Limaria hians*), it occurred as a direct result of pressure removal (Trigg and Moore, 2009). The scale to which pressure removal is required is likely to be species specific, dependent upon the existing spatial extent of the feature and the surrounding habitat potentially suitable for recolonisation and on the desired extent of the feature upon recovery. Most of the species and habitats with a recover or conserve objective are in regions where mobile demersal fishing methods (hydraulic dredging, otter trawling and scallop dredging) are permitted. Given the degree of physical disturbance associated with these activities, it is likely that any recovery of benthic species will be limited unless these physical pressures are removed from the immediate vicinity of the species or habitat of interest. Furthermore, if these species and habitats are reliant on propagules from populations, it may also be necessary to protect these source areas and hence pressure removal may have to be applied over a greater area. Based on the limited potential for recovery and restoration (Section 4.3.1) for species such as maerl, the main option for maintaining these species stems from their initial protection.

#### 5.2.1.2 Hydrodynamic regime in relation to indirect (far-field) pressures

Assessment of the hydrodynamic regime in the surrounding area will provide an indication of the likely influence of activities and pressures, and the management requirements, in the surrounding area. That is, activities which are not necessarily taking place within the immediate vicinity of a feature, but result in significant sediment re-suspension and redistribution, are likely to impact upon features of conservation interest. This particularly applies to maerl where increased siltation blocks the interstitial spaces, impacting upon the integrity of the associated community. Similarly, seagrasses are highly sensitive to increased turbidity and siltation. Where the local current patterns are such that re-suspended sediment is likely to be transported and deposited in the area of such a feature, pressure removal or management to a distance or location whereby their influence is removed will increase the potential for recovery.

#### 5.2.2 Size of the management area for MPA features

In general, MPAs must be large enough to protect all relevant species, habitats and ecological processes occurring within their boundaries, including movement of mobile species. These principles are transferrable to the management area required for adequate protection and are considered particularly relevant to the way in which the future use of a

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<sup>4</sup> Available for each site from: <http://www.snh.gov.uk/protecting-scotlands-nature/protected-areas/national-designations/mpa-consultations/>

designated area by species may change over time. MPA size, or the size of the management area within MPAs, is often a trade off between economic viability, user conflict, practicalities of enforcement and monitoring and local support (McLeod *et al.*, 2009). Whilst large size is recommended (McLeod *et al.*, 2009; Angelini and Silliman, 2012), the physical and biological processes occurring within and between the MPA features (and those outside of the MPA boundaries) must be considered. McLeod *et al.* (2009) stated that straight boundaries and simple, rather than convoluted shapes reduced the perimeter to surface area ratio, thus reducing what are termed 'edge effects'<sup>5</sup> and thereby increasing the effectiveness of the protection measures. It is emphasised that whilst many authors (e.g. McLeod *et al.*, 2009; Angelini and Silliman, 2012) make recommendations about MPA size and design, these factors are likely to be site and species or habitat specific.

The scale of management will depend, to a large extent, on the feature status across the range of MPAs in the network, coupled with the conservation obligations and aims, and on the degree of reliance of sub-populations on each other. If all replicates of a given feature are in favourable condition across all MPAs except one then management for recovery would logically only be focused on the single MPA. It is also heavily dependent on the specifics of the network, especially in terms of effective physical connectivity mediated through prevailing currents and barriers to species movement. In defining the area over which management should be applied, Williams *et al.* (2010) highlighted the importance of understanding the spatial scales at which physical drivers of community structure operate. That is, how is species distribution influenced by wide scale processes such as latitude and temperature regime and localised factors such as substratum.

Whilst it is not possible to calculate the area requiring management or protection for all possible situations, understanding the underlying principles that dictate habitat occupation, dispersal and reaction to pressure may provide an insight into the relative spatial area that requires management or protection for the various features, at a generic level. An approach is proposed here that will enable an assessment of the value of increasing or decreasing the area of protection or management, in terms of the influence on recovery potential. This approach is based on a number of key influential life history traits, coupled with an assessment of the level of pressure. Such life history traits include adult mobility, reproductive strategy, larval survival, longevity, number of habitats used throughout life history, fragmentation of species and habitats, occupation of potential habitat, degree of incidence of pressure and severity of pressures. The combination of these variables greatly influences the benefit likely to be derived from the nature and extent of management measures implemented for site protection, and how close the management boundaries have to be in relation to the current and potential footprint of the existing feature. As such, nine factors have been identified, described in full in sections 5.2.2.1 – 5.2.2.9 and summarised in Table 5, which is referred to as the management boundary tool from this point forward.

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<sup>5</sup> Where seabed communities lie at the edge of a protected area they might not be buffered particularly well from the diffuse pressures that exist just outside the boundary of the managed area. This reduction in an effective spatial buffer between communities and sources of local pressures (which may be biotic or anthropogenic) often results in reduced community condition. Compact MPAs maximize interior area, diminish 'edge-effects', and reduce the loss of protected species across borders through migration.



Pressures	Relative size and overlap	Low  Isolated point source pressure with ~10 % overlap  +	Medium  Site-scale pressure with ~50 % overlap	High  Sub-regional pressure with ~75 % overlap	Very high  Regional pressure with ~100 % overlap	Not known/Not relevant
	Severity	Low  Small, sub-lethal effects	Medium  Large sub-lethal effects and some mortalities (~10 %)	High  Moderate mortality rate (~50 %)	Very high  High mortality (>75 %)	Not known/Not relevant
					+	

### 5.2.2.1 Adult mobility in relation to the relative size of management boundaries

Adult mobility relates to the movement potential of the species of interest. Adult mobility is considered on a gradient from sessile species (relatively stationary since post-settlement: 0 – 10 m movement), through species with limited mobility (10 – 100 m) along the seabed to highly mobile species (>100 km). Sessile species would include the majority of the benthic epifaunal species, burrowing infaunal species and macrophytes (macroalgae and angiosperms). Once these species settle onto the seabed, they subsequently remain at or very close to their original settlement location. For the purpose of this assessment, sessile species relate to benthic invertebrates with limited potential for movement within the 1 – 10 metres range. In contrast, free swimming pelagic species are considered to be highly mobile with movement ranges measured in kilometres. Adult mobility therefore has a clear influence on the setting of management boundaries. Closely fitted boundaries can be set around limited adult mobility?, sessile species (e.g. maerl), whereas boundaries for highly mobile species, (e.g. skate) require boundaries that allow for continuous dispersal and as such need to be set much wider around distributional observations.

### 5.2.2.2 Reproductive strategy (larval mobility) in relation to the relative size of management boundaries

The reproductive strategy of the species is included as a representation of the mobility of the larval phase. This gradient covers vegetative reproduction (no larval/planktonic dispersal phase and hence the smallest possible larval/juvenile dispersal) to lecithotrophic species (limited planktonic residence and hence pelagic dispersal) and planktotrophic species (longer planktonic residence and potentially the greatest dispersal). Again, with larval/early life history phase movement being absent or highly restricted, it would seem appropriate to fit management boundaries closer to that of the adult occupation footprint. By contrast, species that disperse larval widely may need broader and more extensive levels of spatial protection to maintain overall population sustainability.

### 5.2.2.3 Larval survival and the relative size of management boundaries

Larval survival collectively covers the factors influencing the mortality of pre-settlement and early post-settlement stages. Where survival is high, it is likely that the exact shape and coverage of the managed area is relatively unimportant. Large management boundaries relative to the size of the species/habitat distribution are unnecessary and cannot be justified, however, when initial survival is reduced, as has been observed in *M. modiolus*

subpopulations (Comely, 1978), it is likely that areas of settlement will need to be carefully identified and managed appropriately. This may subsequently require the protection of larger spatial areas to ensure enough juveniles are protected and ensure sufficient recruitment to the population. This protection could be in the form of large and expansive boundaries around settlement areas or through a series of management areas around the source populations which are connected to the population in the MPA. Often these areas of settlement are associated with the presence of adults, hence efforts to improve settlement success are inextricably associated with the preservation of adults. Due to the diffusion of larvae and spatial spread of settlement, protective boundaries will have to be large relative to the size of the feature (right-hand columns in Table 6) if management aims to influence the survival of post-settlement phases.

#### 5.2.2.4 Generation times and the relative size of management boundaries

Generation time refers to life span and is related to the long term expression of the three factors above. Species with a short generation time will rapidly undergo several life history interactions in a short period of time. Based on the potential for dispersal within the larval and adult phase, the shorter generation times are likely to be associated with more frequent shifts in feature extents (as slight shifts in extent between generations are magnified over time). Species with inherently higher rates of dispersal, and hence lower spatial permanency, are likely to benefit less from small and close-fitting management boundaries placed around existing feature footprints. Conversely, long-lived species are likely to exhibit higher spatial fidelity or predictability over the lifetime of management practices and these should be attributed in left-hand columns in Table 6.

#### 5.2.2.5 Number of habitats occupied and the relative size of management boundaries

The number of habitats used during the life cycle of a species provides another important consideration for the selection of protected areas. Species requiring more habitats for the completion of their life cycle are likely to require greater spatial area protecting. The use of multiple habitats in this sense is similar to the identification of Essential Fish Habitat for fisheries species routinely undertaken within the United States of America (Rosenberg *et al.*, 2000). Species occupying the same habitat (i.e. a species which does not require different habitats for spawning, larval and juvenile development, adult life and feeding) throughout its life history is not likely to be overly mobile and may therefore be well managed within more closely-fitting boundaries. Species using more than one habitat, e.g. adult herring that spawn on benthic gravels with subsequent larval phases within the plankton and adult foraging within the wider pelagic habitat, will have more varied distributions, with seasonal occupation, and also require the routes between these habitats to remain accessible. Therefore, there is a management need to set large defined boundaries when species use a greater number of habitats during their life history. The larger area may either be in the form of one large area or multiple areas incorporating connecting habitat. Placement of smaller boundaries and areas based on the current extent or habitat of occupation is not likely to include currently unoccupied, yet essential, habitats required for completion of all life history phases.

#### 5.2.2.6 Fragmentation and the relative size of management boundaries

Fragmentation refers to the patch dynamics within the population (including source/sink<sup>6</sup> population considerations). A highly fragmented habitat will be patchy, with individual patches/sub-populations potentially showing high degrees of isolation. With high levels of

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<sup>6</sup> Source populations are net providers to the recruitment of spatially separated populations. Sink populations, in isolation, cannot sustain a population and require the input of propagules from source populations.

fragmentation, two critical spatial issues need to be considered. Firstly, the level of connectivity between sink and source populations, e.g. are sink populations still connected to a sufficiently sized source population. Secondly, if the fragmentation is a dynamic processes, e.g. within seagrass beds, then new patches have the potential to occur anywhere within the potential habitat range. Both of these issues require the management boundaries to be relatively large in comparison to the feature footprint. Smaller or closely fitting boundaries are (i) not likely to protect the 'meta-population' dynamics (i.e. the interaction separate populations of the same species) which often support fragmented populations or (ii) allow for the dynamic occurrence of new patches outside the current area of occupation. As such, the level of fragmentation relates to the relative size of the management boundaries, with highly fragmented populations requiring proportionally larger boundaries (right-hand columns of Table 6). Habitats and species with a cohesive extent are likely to be adequately protected by smaller and closely fitted management boundaries (left-hand columns of Table 6).

#### 5.2.2.7 Occupation of potential habitat and the relative size of management boundaries

Unlike fragmentation, the area of potential habitat occupation is concerned with the relative overall size of habitat or population extent rather than spatial dynamics within the distribution. Potential habitats that have a low level of occupation by species (and other habitat types), are likely to be small, and may be considered rare or depleted. Conversely, where habitats and species fully occupy the potential habitat available to them, these habitats may be considered 'healthier'. As such, the relative size of management boundaries in comparison to the current feature footprint does not have to be particularly big. However, for features that only partially occupy their potential habitat, the boundaries must be sufficiently large enough to allow for expansion into new areas. Based on this, features with a high degree of habitat occupation do not need large management boundaries relative to the current footprint and are scored attributed in left-hand columns of Table 6.

#### 5.2.2.8 The size of the pressure footprint and the relative size and overlap with the management boundaries

The degree of pressure incidence is a relative estimation of the degree of spatial overlap between the feature of interest and a pressure footprint. For example, the commercial extraction of maerl has a very high degree of incidence with maerl beds. Conversely, fishing activity may adjoin maerl beds, generating only a small area of feature/pressure overlap. When the footprint of the pressure is small and localised within the distribution of a species, it is perhaps unnecessary to create large and generous management boundaries relative to the current footprint of the feature, provided that the feature is protected from the wider influences of the pressure. Conversely, diffuse and widely distributed pressures that overlap a substantial proportion of the feature are likely to require proportionally larger management boundaries. As such, situations with a high degree of overlap between pressures and features will require large management boundaries and would be attributed with far right-hand columns. Low incidence situations are attributed in left-hand columns (Table 6).

#### 5.2.2.9 Pressure severity and the relative size of management boundaries

Pressure severity (including direct (near field) and indirect (far field pressure, as noted in sections 5.2.1.1 and 5.2.1.2) has also been included as an important criterion influencing the size and fit of management boundaries. The relationship with the relative size of the management boundaries relates to the likely repercussions of leaving a particular pressure unmanaged. If the impact of a pressure is considered to be severe, then areas subjected to this pressure are likely to be significantly damaged. Due to the likely severity and subsequent loss, management boundaries need to be relatively large in comparison to the feature footprint as the repercussions of allowing a severe impact anywhere within the feature extent will undoubtedly result in significant damage. Weak pressures may generate

lesser impacts which might be considered more acceptable. Large and all encompassing management boundaries in response to a mild pressure are likely to be considered excessive and garner little support. As such, features subject to severe pressure are attributed with right-hand columns in Table 6) whereas mild pressure situations are attributed with left-hand columns. This section emphasises that the management area must be proportional to the extent and severity of the pressure and to the degree of damage and that unnecessarily large management areas cannot be justified.

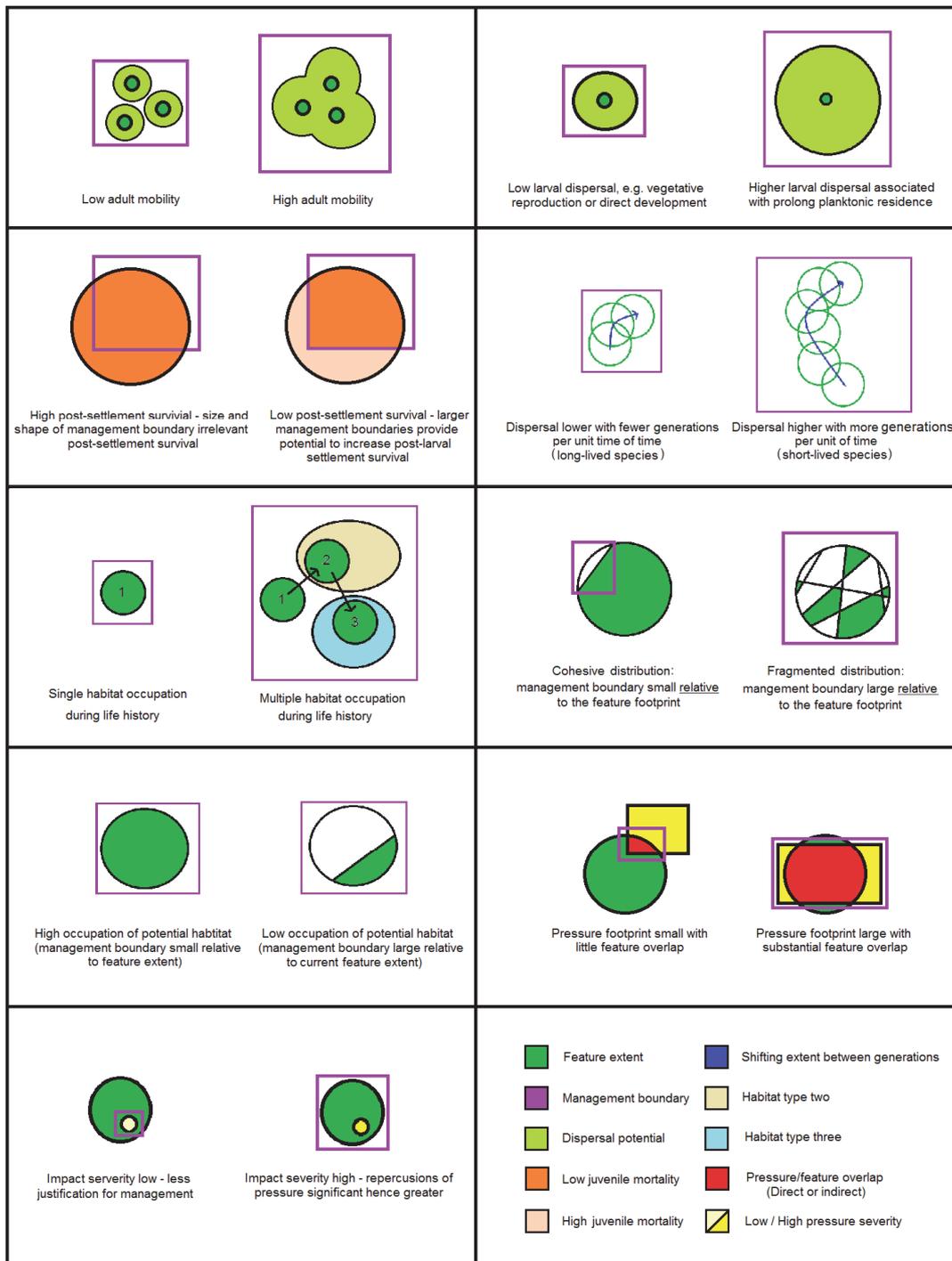


Figure 4. Diagrammatical representation of the factors that influence the size and shape of management boundaries. Pressure footprints represent the direct and indirect pressure sources.

#### 5.2.2.10 Assessment of results from the management boundary tool

Essentially, this approach provides an indication of the relative value of increasing the spatial extent of protection relative to the extent/distribution of the feature in terms of increasing recovery potential. It is emphasised that assessment should be on a site and feature-specific basis. By considering the results obtained across the nine factors, site-specific assessments for individual features can be undertaken, providing a indication of the relative extent of the spatial area required for protection. Features which exhibit more elements/traits which fall to the left hand side of the scale in Table 6, will benefit from relatively small and well-fitted management boundaries in relation to the feature's footprint/distribution, with stringent controls on damaging activities within the site. This may apply to features whose potential to recover or expand may take a long time e.g. hundreds of years in the case of maerl, but that require protection from further decline. The management area must be large enough to protect the feature from pressures beyond the boundary (e.g. to prevent excessive sedimentation arising from sediment resuspension associated with the use of mobile demersal fishing gear). This could mean a closely-fitted boundary to the current observed footprint of the feature of interest (accounting for direct and indirect pressure). Potential habitat outside the area occupied is less likely to be recolonized and management is best focused on the remaining observed feature. Appropriate management may include stringent site protection and the elimination of the relevant overlapping pressures.

Where a feature is exhibiting more elements/traits that fall to the right hand side of the scale in Table 6, this suggests a dynamic and often widely distributed feature. Recruitment may be spatially sporadic. Potential habitat outside the area currently occupied is highly likely to be colonised in the future and the existing footprint may only be a temporal state. Such features will benefit less from closely fitted management boundaries and small-scale pressure management. The management boundaries should be significantly larger relative to the current observed extent of the feature, i.e. for sedentary benthic features the boundaries are buffered significantly beyond the feature footprint. This would ensure protection of the existing feature from influences outside of the management area and/or would enable expansion of the feature if the habitat was suitable and if the feature had the potential to recover. Additionally, this approach would ensure protection of all relevant areas used by mobile species, to encompass e.g. spawning and breeding grounds, adult habitat and migration routes. This may include delineating the entire extent, even potential habitat extent or theoretical distribution. Restoration management of such features should operate at the 'potential habitat' scale and may involve management of areas currently unoccupied by the feature of interest.

The output of this tool is not quantitative and is only indicative of the relative size of the managed area in relation to the known distribution/extent of the feature. The management boundary tool can be applied to species within an individual MPA or applied to the same species for multiple sites. Equally, the tool is not limited to a particular group of species (e.g. those of conservation value) but applicable to all features. The tool also includes consideration of responses to pressures, hence allowing site-specific issues to be considered. This tool could be developed further by considering the weighting of the biological factors according to ecological value. A confidence assessment should also be undertaken as part of the factor assessment presented above and in Table 6. An example product of the confidence assessment is illustrated in Table 7.

A hypothetical example is presented in Figure 5. It cannot be stressed enough that the outcome is an indicator of the value of *relative* size of management boundaries when compared with the feature. Management boundaries can be set inside the feature, covering the entirety of the feature or even substantial areas round the current distribution (Figure 5). It does not provide any indication of the actual spatial area over which management and/or protection should be applied, rather it is indicative of the relative value of increasing the area

of management/protection beyond the immediate footprint of the feature, based on aspects of its biology and the influence of pressures. In this example, the value of increasing the management area beyond the spatial footprint of an existing maerl bed is low compared to that of *M. modiolus*, despite the suitability of the surrounding habitat (and potential for colonisation), because of its low dispersal ability and extremely slow growth. In contrast, the larval dispersal ability of *M. modiolus* is high which increases the potential for larval exchange between populations or sub-populations, thus increasing the potential value of relatively larger managed areas. By comparison, skate is a mobile species which uses different habitats in the adult and juvenile life stages, indicating that a management area should be significantly greater relative to the current extent of the distribution of individuals at any one moment. As indicated in Section 5.2.1., the direct (near-field) and indirect (far-field) pressures should be considered when making any assessment of the spatial extent of the management/protection area. Figure 5 also shows the random attribution used in Tables 5 and 6.

*Table 6. Confidence assessment associated with the assessment matrix. All factors are scored the same as Table 6 ((random attribution (Figure 5)).*

	<b>Certainty</b>			
Adult mobility	Not known +	Low	Medium	High
Reproductive strategy	Not known +	Low	Medium	High
Larval survival	Not known +	Low	Medium	High
Generation time	Not known +	Low	Medium	High
Number of habitats used during the life history	Not known	Low	Medium +	High
Fragmentation	Not known	Low	Medium	High +
Occupation of potential habitat	Not known	Low	Medium +	High
Degree of incidence	Not known	Low +	Medium	High
Severity	Not known	Low +	Medium	High

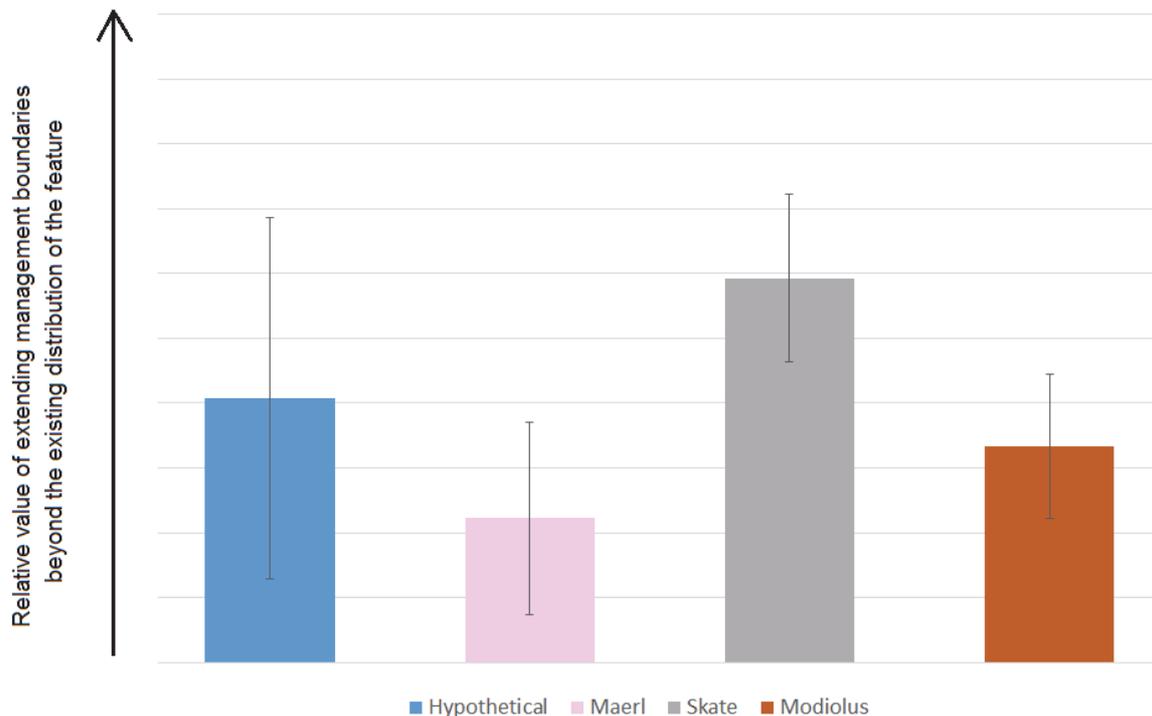


Figure 5. Example of scoring features using the matrix to assess the relative value of extending management boundaries beyond the existing feature's footprint. Features which have columns at the bottom end of the y-axis, are those where there is no or limited value in extending the management boundaries for a feature beyond the feature's extent. Features that reach the top end of the scale are those for which there would be the the greatest benefit to having the entire potential extent of the feature protected within management boundaries. The hypothetical column represents an evaluation of the scoring from Table 6 .

### 5.3 Active intervention and restoration to improve recovery potential

#### 5.3.1 Habitat enhancement

There are few direct studies on habitat enhancement for the species and habitats under consideration (exceptions include Roberts *et al.* (2011) for *M. modiolus*, Boese *et al.* (2009) for *Zostera* spp.). Some techniques are well developed for the restoration of dredging and depositional sites, such as (i) the use of dredging to translocate surplus material, (ii) capping (covering contaminated sediments) and (iii) 'bed levelling' to remove bulk dredging scars. Whilst this approach is very effective in the context of maintenance dredging, where water depths are relatively shallow and sediments are comparatively soft, its effectiveness in the typically deeper water and coarser sediments of aggregate extraction areas is largely unproven. Equally, the activities requiring these remediation techniques are not present in Scottish waters.

Small-scale restoration experiments have shown that it may be possible to address, to a greater or lesser extent, some of the residual physical impacts left by extraction activities. For example, gravel seeding could be used to restore the composition of seabed sediments in areas characterised by an overburden of sands following extraction or sediment screening. Cooper *et al.* (2011) indicate that, in areas of gravel extraction in the North Sea, gravel seeding has successfully been used as a means of restoring the physical characteristics of the substratum, followed by a degree of recovery of the benthic community. Whilst full recovery was not achieved (partly due to the short-term nature of the

study in comparison with the timescales required for recovery) the method did demonstrate sufficient value as a restoration method.

However, Cooper *et al.* (2013) undertook an economic analysis of site restoration using a mixture of techniques. 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.

### 5.3.2 Translocation

In bivalve species, there is limited evidence that translocation may have potential to enhance recovery. For example, Wu and Shin (1998) successfully transplanted individuals of *Pinna bicolor* (a species closely related to *Atrina fragilis*) and found the highest rates of survival and growth to occur in enclosed cages or on rafts (compared to the open seabed) where the bivalves were less accessible to predators, particularly fish and crabs. However, this was a short-term study (12 months) and long-term survival and reproductive potential was not assessed. Furthermore, this study gave no indication of potential for survival and long-term recovery once the bivalves were translocated from the enclosures or rafts, to their natural habitat. Solandt (2003) provides anecdotal evidence that two individuals of *A. fragilis* were successfully transplanted and whilst there is no information on the potential for reproduction, population growth or long-term survival, this, together with the study by Wu and Shin (1998), indicates that translocation experiments for this species would be worth further investigation. However, such experiments should be approached cautiously, with careful selection of the source population and the transplantation location since the integrity of existing populations should not be compromised.

Successful transplantation has also been achieved in the bivalves *Pinna rugosa*, *Crassostrea virginica*, *Argopecten irradians*, *Mytilus edulis*, *Mya arenaria* and *Mercenaria mercenaria* where success was dependent upon optimum stock density and animal size, water temperature, current regime and minimising disease and predation (Wu and Shin, 1998). These factors should be considered in any transplantation experiments for the bivalve species of relevance to this study. Joaquim *et al.* (2008) indicated that, in combination with management measures to prevent losses, transplantation of *Spisula solida*, as a bivalve stock enhancement strategy, could potentially enable recovery of an economically viable fishery. It is of note that the survival rate of this species following transplantation, combined with the relatively short time taken to reach sexual maturity, indicates potential for spawning. Any transplantation effort would need to consider predation, density dependent effects, individual size and other pressures which may influence bivalve survival.

Roberts *et al.* (2011) indicated that natural recovery of *Modiolus modiolus* in Strangford Lough (Northern Ireland) was unlikely. However, Fariñas-Franco *et al.* (2013) successfully transplanted clumps of *M. modiolus* and found higher levels of spat settlement to be associated with transplanted mussels than on cultch (using *Pecten maximus* shell) or natural reefs. Spat densities on translocated populations ranged from 100-200 individuals m<sup>-2</sup> compared to 4-52 spat m<sup>-2</sup> on cultch and 8-36 spat m<sup>-2</sup> on natural reefs. Furthermore, 223 taxa were recorded 12 months after placement of cultch mounds indicating potential for recovery of the associated community as well as for the species. Given the longevity of *M. modiolus* and the time taken to reach sexual maturity, this study does not provide any indication of the long term success of restoration efforts. It does, however, indicate that further, longer term experimental work may be worthwhile.

There is evidence for the potential success of transplantation of *Ostrea edulis* in Strangford Lough (Roberts *et al.*, 2005) although overall restoration failed as a result of unregulated

fishing activities. The translocation efforts were part of an aquaculture effort to increase oyster production locally. Although the 'commercial' brood stock was only a temporary addition to the lough, the spill-over of released gametes resulted in a significant increase in the wild oyster beds. These increases were unfortunately offset through illegal removal of the mostly intertidal wild beds. However, this study does demonstrate that commercial stocks are not cultured in isolation of wild populations but may in fact contribute significantly to them. The co-location of cultured oysters must, however, be balanced against the associated high risk of disease introduction from imported oyster (indeed even differing oyster species) to the wild beds.

Whilst *Limaria hians* has shown high potential for recovery (albeit over a long time period) following small-scale disturbance, translocation of this species may be worth investigation where larger-scale disturbance has taken place and recovery is slow or not evident. It is emphasised that no studies were found to indicate that this approach would be successful for *L. hians*. It is of note that attempts to restore bivalve populations have frequently been unsuccessful because the cause of the decline has not been addressed, coupled with inappropriate habitat conditions, low population density and life history traits that automatically restrict recovery potential (Joaquim *et al.*, 2008).

Another useful example of translocation-based restoration efforts come from the numerous attempts to reintroduce *Zostera* spp. to previously occupied ranges. For *Zostera*, there is no evidence that transplantation works; seedling survival is generally poor following transplantation (Boese *et al.*, 2009). *Zostera* recovery is a function of restoring the nutrient and suspended sediment concentrations (and therefore light availability) and the particle size distribution and organic carbon content of the sediment (Campbell and McKenzie, 2004). Sediment nutrient concentrations are also likely to be a factor. Cabaço *et al.* (2008) stated that whilst *Zostera* tends to recover from natural disturbance events, recovery following anthropogenic disturbance can be limited due to permanent changes to the sediment with long-lasting, negative effects on the *Zostera*. An example of this is scouring following anchoring, where the scour pits fill up with coarser sediment/shell fragments and are not suitable for colonisation. *Zostera* colonisation is exclusively by rhizome growth so recovery is very much dependent upon the size of the disturbed area and the availability of remnant plants/patches (Boese *et al.*, 2009).

### 5.3.3 Hatchery cultivation, sea ranching and stocking enhancement

Aquaculture rearing techniques are increasingly being used to supplement both commercial, wild-caught fisheries and the restoration of conservation species. Of 250 species of marine animals cultured for food by 2007, 97% were brought into culture in the past century and about 20% in the past decade (Duarte *et al.*, 2007). This demonstrates the recent development of aquaculture techniques and their viability in supporting high volume and commercial-scale ventures. This also highlights the potential for aquaculture-based cultivation methods to be harnessed for fisheries enhancement and restoration initiatives globally e.g. the adoption of *C. virginica* cultivation for oyster reef restoration in Chesapeake Bay (Schulte *et al.*, 2009).

Gardner *et al.* (2013) examined ten years of data on catches of stocked and wild *Acanthopagrus butcheri* (a sparid) in an Australian estuary post-restocking and demonstrated that restocking has resulted in a substantial contribution of stocked fish to commercial catches of the species, while eliciting a compensatory growth response in wild fish. Hamasaki and Kitada (2013) analysed long-term data to estimate impacts of climatic factors and hatchery stocking on *Penaeus japonicus* (kuruma prawn) catches, concluding that climate was an important driver of kuruma prawn catches. While contributions of the hatchery programme to fisheries catches had been low, stocking has the potential to stabilize catches in the face of wild population decline due to climate change. In an extensive

historical review of salmon ranching in Japan, Miyakoshi *et al.* (2013) concluded that the hatchery programmes have been successful in increasing commercial catches while conserving genetic diversity locally. The success in re-stocking of fish populations indicates that this technique may be applicable to bivalves and there is evidence of this in the oyster fishery (e.g. see Kennedy and Roberts (2006) for *O. edulis* and Schulte *et al.*, 2009 for *C. virginica*).

Dolorosa *et al.* (2013) evaluated the potential for restoring overexploited stocks of *Trochus* (topshell) by translocating wild animals and rearing hatchery animals to a larger release size. They conclude that the translocation of wild *Trochus* into a network of marine reserves has potential for restoring populations. The value of hatchery-based augmentation of wild populations has also been demonstrated for abalone and harvested sea cucumber species in the Philippines (Juinio-Menez *et al.*, 2013; Lebata-Ramos *et al.*, 2013).

The *Modiolus* Restoration Research Project formed to assist in the recovery of Strangford Lough *Modiolus* biogenic reef also investigated the use of hatchery techniques. It was found to be possible to undertake the full larval cycle from fertilized eggs to settled pediveligers within laboratory conditions. Spat were successfully reared to 1.5 mm after four months in a culture system. Roberts *et al.* (2011) concluded, however, that there are several obstacles to producing enough quantities of spat for reseeded. These were 1) lengthy developmental cycle; 2) slow larval and spat growth; 3) poor survival rates; and 4) very specific settlement requirements. The high costs associated with running the hatchery operations compared to the poor return in seed means hatchery production of *M. modiolus* was not considered a viable restoration option for this species in Strangford Lough.

Although hatchery and stock enhancement techniques were not appropriate for restoration activities in Strangford Lough due to the quantities of individuals required, it is likely that these methods will be of increasing value as a restoration tool for other features such as *L. hians*, *A. islandica* and *A. fragilis*, i.e. where the required restoration densities might be less than for species generating dense, biogenic reef structures. As these approaches are increasingly being adopted for commercial species, the techniques are likely to improve rapidly, thereby increasing their value for application in conservation restoration efforts for many other species.

#### **Box 16. Conclusions regarding the factors influencing recovery**

- The main factors influencing recovery, identified 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, were described in Section 5.1.
- Based on these factors, a 'management boundary tool' has been constructed to guide the relative size of the management boundaries encompassing recovering features.
- Section 5.3 provided an overview of the common, and currently most effective, restoration methods. Currently, the methods derived from aquaculture are the most developed and effective. Other methods require development to improve feasibility, cost-effectiveness and performance.

## 6. KNOWLEDGE GAPS

At a generic level, two significant knowledge gaps are apparent. Firstly, recovery of the species under consideration have generally not been studied. In some cases, examples from related species and other habitats with broadly similar physical characteristics have been the only source of information providing an indication of recovery potential (i.e. the use of *C. virginica* studies for *O. edulis* and *Posidonia oceanica* for *Zostera* spp.). Where recovery has been studied, the timescale for recovery vastly exceeds the timescale of the study (frequently <1 year to 2 years) and therefore, whilst they provide an indication that recovery may be possible for a particular species or habitat, they do not provide any indication of whether or not recovery will continue to ultimately result in a self-sustaining population or community. For most of the species under consideration, no recovery has been documented. As such, there is an urgent requirement for the commissioning of long-term studies to examine the time-scale and trajectory of both passive (pressure abatement only) and active (implementing restoration activities) recovery programmes for several species and habitats. It is likely that this is particularly feasible for the species with shorter life-spans/generation times, e.g. *L. hians*, *A. fragilis*, *O. edulis* and *Z. marina*.

Furthermore, knowledge of the biology of many of the species with a recover objective is lacking. Important information regarding fecundity, dispersal capability, allee effects in specific species and populations is lacking. Equally, growth rates and age-specific mortalities (life tables) are not documented for many species. This information allows estimates to be made regarding the potential for meta-population connectivity (dispersal between fragmented sub-populations), allee effect thresholds and appropriate monitoring for establishing the trajectory and long-term stability of recovery. Only with this information can decisions be made about whether passive recovery can be expected and when active restoration might be required. It is therefore suggested that a database is constructed containing a list of critical autecological attributes for the species considered here (or the main species comprising the habitats considered above). This will identify, of each species, which properties will do not current have information for, e.g. fecundity, dispersal, natural mortality (life table analysis) and exact habitat requirements. Once these parameters have been identified, further work will have to be commissioned to either observed or modelled to generate the required values.

Discussions during the workshop (Annex 3) highlighted significant gaps in the knowledge of the distribution and condition of some of the species, communities and habitats with a recover objective. Where these features are known to exist, there is no knowledge of whether abundance and diversity is typical or whether it would be improved through better management and pressure removal. As such, there is also a requirement for enhanced site-scale assessments of remaining features so that initial recovery potential can be better assessed. This should include aspects of (i) current density and extent, (ii) habitat suitability, (iii) connectivity to other local populations of the same species or habitat, (iv) pressure intensity experienced at the site, (v) current population/habitat trajectory and (vi) the quality of the associated assemblage for biogenic features. This information provides a better knowledge base both for the estimation of recovery potential and the generation of an appropriate recovered reference point.

At present, the appropriate spatial scale of management cannot be objectively defined with a set of principles, and a site-specific basis needs to be adopted. An approach has been proposed but this will need to be tested on a feature and site-specific basis.

Finally, the repertoire of active restoration techniques is limited in the marine environment and such techniques are very much in their infancy when compared to terrestrial methods. Further work is required to investigate novel strategies for encouraging marine recovery in both species and habitats. The most productive lines of inquiry are likely to include (i) the

potential for aquaculture techniques to be applied to species of conservation importance, (ii) partial translocation of propagules, juveniles or adults to establish new areas, move endangered features or improve overall connectivity between populations and (iii) habitat enhancement (e.g. increased use of cultch, predator control, artificial reefs etc.) to improve settlement, recruitment and overall survival.

**Box 17. Conclusions regarding the presence of knowledge gaps within the recovery literature**

Knowledge gaps were apparent for the majority of species and habitats considered in this report. Gaps were apparent for the basic biological and ecological properties for species species. Furthermore, rarely had the recovery process for any feature been quantitatively monitored to a 'recovered' point. Site-specific information for establishing current trajectory and initial condition was also highlighted as often being poorly documented.

## 7. CONCLUSIONS AND RECOMMENDATIONS

A review of the literature indicated a high degree of variation in the use and interpretation of the term 'recovery'. It is proposed that 'recovery' should refer to a process or trajectory and 'recovered' should refer to an end point:

**Recovery** is considered as:

- *a consistent trajectory, detectable above natural variability, of a representative set of feature properties from a previous (or otherwise defined) state towards the Recovered Reference Range, throughout a spatially explicit area.*

**Recovered** is considered to be a:

- *stable, long-term similarity of a representative set of feature properties of a recovering feature to the Recovered Reference Range throughout a spatially explicit area.*

Separate definitions of recovery and recovered have been proposed for individual species, communities (including biogenic species) and habitats. Targets for achieving recovered status should be based on a number of parameters that contribute to long term stability and targets should be within a range to account for natural variability within both the recovering and reference populations or communities. For long-lived species, this range may require modification over time. Given the longevity of some of the species considered, together with aspects of their biology which reduce their recovery potential, it is suggested that monitoring should be designed to track the recovery trajectory. Reaching recovered status may not be achievable within the lifetime of current policy, conservation strategies and prevailing environmental conditions.

The species and habitats considered in this review are predominantly long-lived, slow growing and either have low fecundity and/or poor dispersal potential and/or poor recruitment. These factors dictate that the recovery process will be slow and, for some species (e.g. species of maerl and potentially some populations of *Modiolus modiolus*), recovery potential is likely to be very limited with the short- to mid-term (Table 7). Furthermore, knowledge of the biology of a number of species is lacking. This is coupled with a lack of pre-existing data on species within MPAs, meaning that targets based on spatial rather than temporal comparisons may need to be used. This means that the development of targets or expectations on recovery rates is problematic.

Of the species and habitats reviewed, recovery potential is variable. Whilst little direct evidence of complete recovery (i.e. achieving recovered status) was found, *Limaria hians*, *Zostera* spp., *Ammodytes* spp and black guillemot all showed signs of recovery following pressure removal. It is emphasised that in some cases (e.g. *Limaria hians*), recovery was studied following small-scale, experimental disturbance and the projected recovery times may be longer following wider-scale, anthropogenic disturbance. In some cases, recovery may not be possible. For other species, no evidence of recovery was found and, due to aspects of the biology of these species, recovery potential is likely to be extremely limited (e.g. *Modiolus modiolus* and maerl)

Table 7. Summary of the recovery potential for species and habitat considered within this report

<b>List 1: species/habitats, supported by evidence, with the potential for recovery</b>	
Timescale: short to medium term	Evidence: moderate evidence base from autecological studies and/or documented recovery case studies
<ul style="list-style-type: none"> <li>• <i>Limaria hians</i> (10 - &gt;100 years) (no assessment of associated community)</li> <li>• <i>Zostera</i> spp (2-7 years, up to 18)</li> <li>• <i>Ammodytes</i> (timescale unclear due to variability in population size. Likely to be tens of years or less)</li> <li>• Black guillemot (approximately 5 years)</li> <li>• Herring spawning ground (5-10 years for physical recovery, 25 years for recolonisation)</li> <li>• Native oyster (<i>Ostrea edulis</i>) (following active restoration)</li> </ul>	
<b>List 2: species/habitats, lacking evidence, but with some potential for recovery</b>	
Timescale: short to medium term	Evidence: Little or no evidence base and lacking sufficient biological information
<ul style="list-style-type: none"> <li>• Maerl or coarse shell with burrowing cucumbers (limited by distribution of maerl but need to investigate potential for recovery of <i>Neopentadactyla mixta</i> in gravel).</li> <li>• Seapens and burrowing megafauna in circalittoral mud</li> <li>• Shallow tide-swept coarse sands with burrowing bivalves (some evidence that some species within this habitat may recover but this depends on current population status. Timescale likely to be tens of years and there is no information on 'community' recovery potential)</li> <li>• <i>Atrina fragilis</i> (some evidence of successful translocation, long-term survival not known)</li> </ul>	
<b>List 3: species/habitats, supported by evidence, lacking the potential for a recovery</b>	
Timescale: no recovery anticipated in the short to medium term although long-term recovery may be possible	Evidence: autecological studies and/or documented recovery
<ul style="list-style-type: none"> <li>• <i>Modiolus modiolus</i> (if possible, likely to take tens or hundreds of years)</li> <li>• Maerl (if possible, likely to take hundreds of years)</li> <li>• <i>Arctica islandica</i> (if possible, likely to take hundreds of years)</li> <li>• Common skate (if possible, recovery period likely to be protracted but may be further limited by low population density and low fecundity)</li> </ul>	

Factors limiting recovery included anthropogenic influence, together with aspects of the biology of individual species and communities and environmental conditions. Of key importance is the scale of the disturbance, the degree of habitat homogenisation/fragmentation and species removal and the longevity of physical modifications to the habitat. Recovery is reliant on environmental and biological connectivity between populations or species patches and on the suitability of the habitat in terms of substratum type, depth, water quality and sediment quality. For some species (particularly those inhabiting soft sediments), subtle change in the sediment structure, consolidation, stability and chemistry are of importance. Superimposed upon this are the life history traits of the individual species, which may or may not enhance recovery potential. These include reproductive strategy and frequency, growth rate, longevity and dispersal ability, coupled with biological interactions between species which may influence the direction and timescale of the recovery trajectory.

Several restoration techniques have been discussed and these appear to be most successful for bivalves (although the success of restoration of biogenic species is unclear), indicating that active intervention to restore some of the species associated with bivalve communities in shallow coarse sands may be worthy of further investigation. For longer-lived, biogenic species, translocation has been successful but there is no evidence of long-term survival or reproductive output. Therefore, these techniques could be trialled but should not replace other forms of management. Other than for some specific aquaculture techniques, the majority of the marine restoration methods are insufficiently developed to be

cost effective and reliable. This, however, should not exclude the use of these methods for highly threatened species and habitats requiring imminent intervention. Furthermore, applying these methods provides the required development urgently needed for restoration practices to become an effective and established management tool.

No studies which examined the influence of different management strategies on recovery potential or recovery timescale were found. However, where recovery was documented (or recovery potential was implied), recovery followed direct pressure removal. An approach to defining a relative spatial scale for management that is feature and site specific has been proposed, based on aspects of the species biology, current status of the habitat and influence of pressures. This indicated that, for species with very low dispersal abilities, slow growth and poor recruitment, protecting an area wider than that of the species footprint may not be as beneficial since it is likely that the species/habitat would either take a very long time or may never expand into it (e.g. maerl bed). In contrast, a species with higher dispersal ability, a shorter life cycle and a higher degree of mobility has the potential to recolonise a much wider area, providing that the habitat is suitable, indicating that wider-scale protection would be beneficial. It should be noted that protection and management should account for direct (near-field) and indirect (far-field) pressures so that, in the case of maerl, protection of an area beyond that occupied by a maerl bed may be necessary to prevent, for example, siltation arising from activities taking place in the surrounding area.

Overall, this review has revealed that of the species, communities and habitats considered, some have very limited recovery potential whilst others are likely to be responsive to pressure removal. Of those with the greatest potential to recover, the timescales for recovery vary but are generally tens to hundreds of years. Whilst habitat suitability and aspects of the species biology are critical factors in determining recovery rate, the scale and nature of the disturbance and their effect on habitat homogenisation, modification, connectivity and fragmentation are of primary importance since these factors dictate the starting point for recovery.

Key conclusions distilled from this report are:

- Clear terminology is required hence definitions for 'recovery', 'recovered' and the 'recovered reference range' have been provided
- A framework has been provided for the assessment of (i) the cause of decline and (ii) the recovery potential. This framework links factors limiting recovery with both passive and active restoration methods.
- The main factors influencing recovery, identified 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, were described in Section 5.1.
- Based on these factors, a 'management boundary tool' has been constructed to guide the relative size of the management boundaries encompassing recovering features.
- Section 5.3 provides an overview of the common, and currently most effective, restoration methods. Currently, the methods derived from aquaculture are the most developed and effective. Other methods require development to improve feasibility, cost-effectiveness and performance.
- Knowledge gaps were apparent for the majority of species and habitats considered in this report. Gaps were apparent for the basic biological and ecological properties for species species. Furthermore, rarely had the recovery process for any feature been quantitatively monitored to a 'recovered' point. Site-specific information for

establishing current trajectory and initial condition was also highlighted as often being poorly documented.

## 7.1 Recommendations

- Declining trajectories must be halted before recovery can occur - where possible, the pressure acting upon species and habitats requiring recovery are immediately reduced. This is particularly important for species which lack the potential for short to medium term recover.
  - Limited knowledge of the distribution and status of some of the features reviewed in this document, together with uncertainties about the activities and pressures leading to their decline, is a significant barrier to implementing effective management to enable recovery. It is suggested that the features most at risk are those which have the lowest recovery potential and which are thought to be influenced by the most significant pressures. As a first, immediate step for features with limited or no documented recovery potential or protracted recovery times (Table 7 – list 3), action must need to be taken to prevent further decline and protect what remains of that feature. The potential for recovery can then be more fully assessed and more detailed management plan can be formulated.
  - For other features, with low risk of further decline (e.g. wider distribution, higher recovery potential), it may be possible to take time to gain a greater understanding of the activities and pressures influencing each feature before management to enable recovery is implemented. However, it is emphasised that where a particular pressure is known to be adversely affecting a feature, it should be removed or controlled in the first instance with further controls being implemented once there is sufficient knowledge (e.g. through research) to formulate and implement a more detailed, optimised management plan.
- The management boundary tool should be consulted when planning the size of site-specific recovery programmes. The location of the management area should be set by influential biological and hydrodynamic processes rather than existing designation boundaries.
  - One aim of this study was to provide an indication of the spatial extent to which management or protection should be applied in order to enable recovery. An approach to establishing the relative value of increasing this area relative to the extent of the feature has been proposed but it is emphasised that this should be applied on a feature and site-specific basis. In particular, SNH would need to identify recovery targets for a feature in terms of spatial extent. Following this, appropriate management areas can be defined.
  - It is emphasised that the management area should be defined according to the influential biological and hydrodynamic processes and not according to MPA boundaries. That is, if the hydrodynamic and biological processes dictate, management should be applied across different MPAs and, if necessary, outside of MPA boundaries to enable recovery. It is emphasised that unless there is replication of and connectivity between the component species and habitats, a high degree of persistence and resilience will not be achieved, regardless of the size of the management area.
- Further work is required to fill knowledge gaps. Priority topics are (i) species specific biological properties (fecundity, growth, survival and dispersal) and demographic connectivity, (ii) restoration methods, (iii) documented recovery trajectories to a recovered end-point and (iv) site-specific assessments of condition and connectivity of features requiring recovery.
  - Improved knowledge of the biology of some of the species (particularly the echinoderms, cnidarians and some of the bivalve species) will enable a better

- understanding of their potential for recovery. There is limited evidence that active restoration work may enhance the recovery potential of some species and whilst it is emphasised that there is no direct evidence that these techniques will lead to recovery, they may be worth investigation. At present, there is insufficient knowledge of the structure of some communities (e.g. burrowed mud) to determine whether or not the community within a particular site is likely to be impacted and whether species composition and density would be enhanced through pressure alleviation. This could be addressed through a review of existing literature and analysis of existing survey data to assess 'typicalness' in different environmental conditions. It may be possible to identify commonalities between communities that are impacted and those that aren't (or those that are impacted to different degrees), in terms of missing species, reduced density or reduced biomass/body size. These may provide a means of assessing the typicalness of communities where the influence of pressures is unknown or unquantified.
- Further research is required to establish both environmental and ecological connectivity between species and habitat patches, and to identify their location in relation to a reproductively active brood stock. Poorly connected or isolated populations need to be assessed for their potential to sustain themselves.
  - Most of the literature relating to MPAs highlights the importance of connectivity, largely in physical/hydrodynamic terms as the potential for propagule dispersal, larval settlement and recruitment. However, habitat heterogeneity and ecological connectivity must be considered, together with the viability of existing populations as a source of new recruits. Therefore, simply assessing the distance between habitat/species patches and the current patterns connecting them may not be sufficient to assess connectivity without some assessment of the biological condition of the existing communities. This may be addressed by further research into the larval source and settlement locations, coupled with hydrodynamic and larval dispersal modelling and genetic studies.
- Recovery programmes consider and protect ecological connecting a recovery site to the wider population, with specific consideration given to the projection of regional source populations.
    - In reality, it may be unrealistic to expect that all examples of a species or habitat can be managed to promote recovery, particularly where habitat patches or populations of individual species are sparsely dispersed over wide areas. In such cases, an area/areas need to be targeted for management (be it pressure removal, pressure amendment or active restoration effort). In addition, populations/habitat patches which have potential to contribute to the overall recovery of the target area need to be considered. That is, certain neighbouring populations may be disproportionately important due to either their size or hydrodynamic connectivity. Identification and protection of these source populations provides the recovering habitat with the propagule pressure required to overcome mortality within the recipient habitat and move towards a recovered state. Similarly, it is possible that the maintenance of a population is reliant on the interaction of various sub-populations within a region and the area of this interaction should be defined. This approach maximises the recovery potential of specified habitat patches/populations whilst not imposing unnecessary management restrictions on other areas. It is emphasised that areas targeted for management must be carefully chosen and identification of the supporting, surrounding populations, must be based on good evidence. Similarly, there must be good evidence for excluding populations that are not considered to be a viable source of propagules and that have little or no potential to recover or maintain their current status.

- For long-term recovery trajectories, 'recovered reference ranges' are routinely updated to reflect environmental change and especially climate change.
  - Subsequent monitoring/assessment studies should be mindful not only of the potentially long recovery periods involved, but also of potential environmental changes that may occur in the interim (so-called exogenic unmanaged pressures). Monitoring strategies should be designed to detect changes that are indicative of recovery (i.e. track the recovery process) as well as detecting the end point of recovery. For some species, the recovery timescale may be so long that recovery end points may be unachievable within the lifetime of current policies, conservation objectives/obligations and prevailing environmental conditions. For this reason, recovery endpoint targets may need to be reconsidered periodically. However, where recovery of an entire community type (e.g. a biogenic species and the associated community or burrowed mud habitats) is the aim, it may be worth highlighting details relating to species of socio-economic value during the monitoring process. Similarly, where recovery of a single species is the aim, it may be worth incorporating measures relating to dependent commercially important species into the monitoring process. This would enhance our understanding of the recovery process at a wider, socially relevant level whilst providing valuable information to feed back to management discussions and stakeholders.

## 8. REFERENCES

- Alexander, R.R., Stanton (Jr), R.J. & Dodd, J.R. 1993. Influence of sediment grain size on the burrowing of bivalves: correlation with distribution and stratigraphic persistence of selected Neogene clams. *Palaios*, **8(3)**, 289-303.
- Allee, W.C. 1931. *Animal aggregations. A study in general sociology*, University of Chicago Press, University of Chicago.
- Angelini, C. & Silliman, B.R. 2012. Patch size-dependent community recovery after massive disturbance. *Ecology*, **93(1)**, 101-110.
- Anwar, N. A., Richardson, C. A., & Seed, R. 1990. Age-determination, growth-rate and population-structure of the horse mussel *Modiolus modiolus*. *Journal of the Marine Biological Association of the United Kingdom*, **70(2)**, 441-457.
- Arnott, S.A. & Ruxton, G.D., 2002. Sandeel recruitment in the North Sea: demographic, climatic and trophic effects. *Marine Ecology Progress Series*, **238**, 199- 210.
- Barbera, C., Bordehore, C., Borg, J. A., Glemarec, M., Grall, J., Hall-Spencer, J. M. & Valle, C. 2003. Conservation and management of northeast Atlantic and Mediterranean maerl beds. *Aquatic Conservation-Marine and Freshwater Ecosystems*, **13**, 65-76.
- Barnes, M. 2008. *Clupea harengus*. Atlantic herring. Marine Life Information Network: Biology and Sensitivity Key Information Sub-programme [on-line]. Plymouth: Marine Biological Association of the United Kingdom. [cited 20/12/2013]. Available at: <http://www.marlin.ac.uk/speciesinformation.php?speciesID=3017>.
- Barnes, K.A. & Conlan, K.E. 2007. Disturbance, colonisation and development of Antarctic benthic communities. *Philosophical Transactions of the Royal Society B*, **362**, 11-38.
- Baxter, J.M., Boyd, I.L., Cox, M., Donald, A.E., Malcolm, S.J., Miles, H., Miller, B. & Moffat, C.F., (Editors), 2011. *Scotland's Marine Atlas: Information for the national marine plan*. Marine Scotland, Edinburgh. pp. 191
- Beare, D., Rijnsdorp, A.D., Blaesberg, M., Damm, U., Egekvist, J., Fock, H., Kloppmann, M., Röckmann, C., Schroeder, A., Schulze, T., Tulp, I., Ulrich, C., van Hal, R., van Kooten, T. & Verweij, M. 2013. Evaluating the effect of fishery closures: Lessons learnt from the Plaice Box, *Journal of Sea Research*, **34**, 49-60
- Bell, S., & Devlin, D.J. 1983. Short-Term Macrofaunal Recolonization of Sediment and Epibenthic Habitats in Tampa Bay, Florida. *Bulletin of Marine Science*, **1**, 102-108.
- Beukema, J.J., Flach, E.C., Dekker, R. & Starink, M. 1999. A long-term study of recovery of the macrozoobenthos on large defaunated plots on a tidal flat in the Wadden Sea. *Journal of Sea Research*, **42**, 235-254.
- Bevilacqua, S., Terlizzi, A., Fraschetti, S, Russo, G.F. & Boero, F. 2006. Mitigating human disturbance: can protection influence trajectories of recovery in benthic assemblages? *Journal of Animal Ecology*, **75**, 908-920.
- BIOMAERL. 1999. *Final Report, BIOMAERL project* (Coordinator: P.G.Moore, University Marine Biological Station, Millport, Scotland), EC Contract No.MAS3-CT95-0020, (in 2 vols), 1: 1–541, 2: 542–973 pp + Appendix.

- BIOMAERL team. 2003. *Conservation and management of NE Atlantic and Mediterranean maerl beds*. *Aquatic Conservation: Marine and Freshwater Ecosystems*, **13**: S65–S76.
- Birkett .D.A., Maggs .C.A. & Dring M.J. 1998. Maerl (volume V). *An overview of dynamic and sensitivity characteristics for conservation management of marine SACs*. Scottish Association for Marine Science. (UK Marine SACs Project). 116 pages.
- Blanchard, M. 1997. Spread of the slipper limpet *Crepidula fornicata* (L. 1758) in Europe. Current state and consequences. *Scientia Marina*, **61(SUPPL. 2)**, 109-118.
- Blaxter, J.H.S. 1990. The herring. *Biologist*, **37**, 27-31.
- Blaxter, J.H.S., & Holliday, F.G.T., 1963. The behaviour and physiology of Herring and other Clupeids. *Advances in Marine Biology*, **1**, 261- 394.
- Boese, B., Kaldy, J.E., Clinton, P.J., Eldridge, P.M. & Folger, C.L. 2009. Recolonization of intertidal *Zostera marina* L. (eelgrass) following experimental shoot removal. *Journal of Experimental Marine Biology and Ecology*, **347**, 69-77.
- Bolam, S.G., Whomersley, P. & Schratzberger, M. 2004. Macrofaunal recolonization on intertidal mudflats: effect of sediment organic and sand content. *Journal of Experimental Marine Biology and Ecology*, **306**, 157-180.
- Bordehore, C., Ramos-Espla, A. A., & Riosmena-Rodriguez, R. 2003. Comparative study of two maerl beds with different otter trawling history, southeast Iberian Peninsula. *Aquatic Conservation-Marine and Freshwater Ecosystems*, **13**, 43-54.
- Borja, Á, Dauer, D.M., Elliott, M. & Simenstad, C.A. 2010. Medium- and Long-term Recovery of Estuarine and Coastal Ecosystems: Patterns, Rates and Restoration Effectiveness. *Estuaries and Coasts*, **33**, 1249–1260.
- Borja, A., Dauer, D.M. & Grémare, A. 2012. The importance of setting targets and reference conditions in assessing marine ecosystem quality. *Ecological Indicators*, **12**, 1–7.
- Bosence, D. W. J. 1976. Ecological studies on two unattached coralline algae from western Ireland. *Palaeontology*, **19**, 365-395.
- Bosence, D. W. 1983. *The occurrence and ecology of recent rhodoliths - a review*. In Coated grains (pp. 225-242). Springer Berlin Heidelberg.
- Bosence, D., & Wilson, J. 2003. Maerl growth, carbonate production rates and accumulation rates in the northeast Atlantic. *Aquatic Conservation-Marine and Freshwater Ecosystems*, **13**, 21-31.
- Boyd, S.E., Cooper, K.M., Limpenny, D.S., Kilbride, R., Rees, H.L., Dearnaley, M.P., Stevenson, J., Meadows, W.J., & Morris, C.D., 2004. *Assessment of the rehabilitation of the seabed following marine aggregate dredging*. Science Series Technical Report 121. Cefas, Lowestoft.
- Boyd, S.E., Limpenny, D.S., Rees, H.L. & Cooper, K.M. 2005. The effects of marine sand and gravel extraction on the macrobenthos at a commercial dredging site (results 6 years post dredging). *ICES Journal of Marine Science*, **62**, 145-162.

- Brown, R. A. 1984. Geographical variations in the reproduction of the horse mussel *Modiolus modiolus* (Mollusca, Bivalvia). *Journal of the Marine Biological Association of the United Kingdom*, **64(4)**, 751-770.
- Brown, R. A., & Seed, R. 1977. *Modiolus modiolus* (L.) - an autecological study. *11th European Marine Biology Symposium*, 93-100.
- Brown, R. A., Seed, R., & Oconnor, R. J. 1976. Comparison of relative growth in *Cerastoderma* (=Cardium) *edule*, *Modiolus modiolus* and *Mytilus edulis* (Mollusca – Bivalvia). *Journal of Zoology*, **179(JUL)**, 297-315.
- Butler, R. G. & Buckley, D.E. 2002. Black Guillemot (*Cepphus grylle*), the birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: <http://bna.birds.cornell.edu/bna/species/675>, doi:10.2173/bna.675.
- Butler, A. J., Vicente, N. and de Gaulejac, B. 1993. Ecology of the pteroid bivalves *Pinna bicolor* Gmelin and *Pinna nobilis* Linnaeus. *Marine Life*, **3**, 37-45.
- Cabaço, S., & Santos, R. 2007. Effects of burial and erosion on the seagrass *Zostera noltii*. *Journal of Experimental Marine Biology and Ecology*, **340**, 204-212.
- Cabaço, S., Santos, R. & Duarte, C.M. 2008. The impact of sediment burial and erosion on seagrasses: a review. *Estuarine Coastal and Shelf Science*, **79**, 354-366.
- Cabioch, J. 1970. The Maerl of the coasts of Brittany and the problem of its survival. *Penn ar Bed (Brest)*, **7(63)**, 421-429.
- Cairns, D. 1980. Nesting Density, Habitat Structure and Human Disturbance as Factors in Black Guillemot Reproduction. *The Wilson Bulletin*, **92**, 352-361.
- Caley M. J., Carr M. H., Hixon M. A., Hughes T. P., Jones G. P., & Menge B. A. 1996. Recruitment and the local dynamics of open marine populations. *Annual Review of Ecological Systems*, **27**, 477–500.
- Campbell, S.J. & McKenzie, L.J. 2004. Flood related loss and recovery of intertidal seagrass meadows in southern Queensland, Australia. *Estuarine Coastal and Shelf Science*, **60**, 477-490.
- Cargnelli, L.M., Griesbach, S.J., Packer, D.B. & Weissberger, E., 1999. *Essential fish habitat source document: ocean quahog, Arctica islandica, life history and habitat characteristics*. NOAA Technical Memorandum, NMFS-NE-148, 12 pp.
- Carroll, M. L., Johnson, B., Henkes, G.A., McMahon, K.W., Vornkov, A., Ambrose Jr., W.G. & Denisenko, S.G. 2009. Bivalves as indicators of environmental variation and potential anthropogenic impacts in the southern Bering Sea. *Marine Pollution Bulletin*, **59(4-7)**, 193-206.
- Carruthers, M., Chaniotis, P.D., Clark, L., Crawford-Avis, O., Gillham, K., Linwood, M., Oates, J., Steel, L., & Wilson, E., 2011. Contribution of existing protected areas to the MPA network and identification of remaining MPA search feature priorities - Maerl or coarse shell gravel with burrowing sea cucumbers. *Internal report produced by Scottish Natural Heritage, the Joint Nature Conservation Committee and Marine Scotland for the Scottish Marine Protected Areas Project*. Available online at: <http://www.scotland.gov.uk/Topics/marine/marine-environment/mpanetwork/epa;> <http://www.snh.gov.uk/docs/B988854.pdf>. Accessed 18/09/2013

CEBC, 2009. *Guidelines for Systematic Review in Conservation and Environmental Management. Version 3.1*. Centre for evidence-based conservation, Bangor University, Wales, UK. Also available online at [http://www.cebc.bangor.ac.uk/Documents/Reviewguidelinesversion3.0\\_FINAL.pdf](http://www.cebc.bangor.ac.uk/Documents/Reviewguidelinesversion3.0_FINAL.pdf).

Chaniotis, P.D., Crawford-Avis, O.T., Cunningham, S., Gillham, K., Tobin, D., Linwood, M. 2011. *Identifying locations considered to be least damaged/more natural in Scotland's seas*. Final report produced by the Joint Nature Conservation Committee, Scottish Natural Heritage and Marine Scotland for the Scottish Marine Protected Areas Project.

Clark, L. 2012. Marine biological survey to establish the distribution and status of fan mussels *Atrina fragilis* and other Marine Protected Area (MPA) search features within the Sound of Canna, Inner Hebrides. *Scottish Natural Heritage Commissioned Report No. 438*, 1-175.

Collie, J. S., Hall, S. J., Kaiser, M. J., & Poiner, I. R. 2000. A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, **69**, 785-798.

Comely, C. A. 1978. *Modiolus modiolus* (L) from Scottish west-coast. 1) Biology. *Ophelia*, **17(2)**, 167-193.

Conan, G. & Shafee, M.S. 1798. Growth and biannual recruitment of the black scallop *Chlamys varia* (L.) in Lanveoc area, Bay of Brest. *Journal of Experimental Marine Biology and Ecology*, **35**, 59-71.

Connor, D.W., Allen, J.H., Golding, N., Howell, K.L., Lieberknecht, L.M., Northen, K.O. & Reker, J.B. 2004. *The Marine Habitat Classification for Britain and Ireland Version 04.05* JNCC, Peterborough. ISBN 1 861 07561 8 (internet version). <http://jncc.defra.gov.uk/page-1584>.

Constantino, R., Gaspar, M.B., Tata-Regala, J., Carvalho, S., Cúrdia, J., Drago, T., Taborda, R. & Monteiro, C.C. 2009. Clam dredging effects and subsequent recovery of benthic communities at different depth ranges. *Marine Environmental Research*, **67**, 88-99.

Cooper, K.M., Barrio Froján, C.R.S., Defew, E., Curtis, M., Fleddum, A., Brooks, L. & Paterson, D.M., 2008. Assessment of ecosystem function following marine aggregate dredging. *Journal of Experimental Marine Biology and Ecology*, **366**, 82-91.

Cooper, K., Burdon, D., Atkins, J.P., Weiss, L., Somerfield, P., Elliott, M., Turner, R.K., Ware, S., 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)**, 33-45.

Cooper, K.M., Ware, S.W., Vanstaen, K. & Barry, J., 2011. Gravel seeding – A suitable technique for restoring the seabed following marine aggregate dredging? *Estuarine, Coastal and Shelf Science*, **91**, 121-132.

Cowen, R. K., & Sponaugle, S. (2009). Larval dispersal and marine population connectivity. *Annual Review of Marine Science*, **1**, 443-466.

Craik, C. 1997. Long-term effects of North American Mink *Mustela vison* on seabirds in western Scotland. *Bird Study*, **44**, 303-309.

- Dernie, K.M., Kaiser, M.J., Richardson, E.A. & Warwick, R.M. 2003. Recovery of soft sediment communities and habitats following physical disturbance. *Journal of Experimental Marine Biology and Ecology*, **285-286**, 415-434.
- De Groot, S.J. 1980. The consequences of marine gravel extraction on the spawning of herring, *Clupea harengus* Linné. *Journal of Fish Biology*, **16**, 605-611.
- De Schweinitz, E. H., & Lutz, R. A. 1976. Larval development of the northern horse mussel, *Modiolus modiolus* (L.), including a comparison with the larvae of *Mytilus edulis* L. as an aid in planktonic identification. *Biological Bulletin*, **150(3)**, 348-360.
- Desprez, M. 2000. Physical and biological impact of marine aggregate extraction along the French coast of the Eastern English Channel: short- and long-term post-dredging restoration. *ICES Journal of Marine Science*, **57**, 1428-1438.
- Dolorosa, R. G., Grant, A. Gill, J. A. Avillanosa, A. L. & Gonzales, B.J. 2013. Indoor and deep subtidal intermediate culture of *Trochus niloticus* for restocking. *Reviews in Fisheries Science*, **21**, 414–423.
- Donovan, D.A., Ellias, J.P. & Baldwin, J. 2004. Swimming behaviour and morphology of the file shell *Limaria fragilis*. *Marine and Freshwater Behaviour and physiology*, **37(1)**, 7-16.
- Duarte, C.M., Borja, A., Carstensen, J., Elliott, M., Krause-Jensen, D. & Marbà, M. 2013. Paradigms in the Recovery of Estuarine and Coastal Ecosystems. *Estuaries and Coasts*. DOI 10.1007/s12237-013-9750-9.
- Duarte, C. M., Marba, N. & Holmer, M. 2007. Rapid domestication of marine species. *Science*, **316**, 382–383.
- Duineveld, G.C.A., Bergman, M.J.N., & Lavaleye, M.S.S. 2007. Effects of an area closed to fisheries on the composition of the benthic fauna in the southern North Sea. *ICES Journal of Marine Science*, **64**, 899-908
- Edwards, D. C. B., & Moore, C. G. (2008). Reproduction in the sea pen *Pennatula phosphorea* (Anthozoa: Pennatulacea) from the west coast of Scotland. *Marine Biology*, **155(3)**, 303-314.
- Edwards, D. C. B., & Moore, C. (2009). Reproduction in the sea pen *Funiculina quadrangularis* (Anthozoa: Pennatulacea) from the west coast of Scotland. *Estuarine, Coastal and Shelf Science*, **82**, 161-168.
- Elliott, M. 2011. Marine science and management means tackling exogenic unmanaged pressures and endogenic managed pressures – a numbered guide. *Marine Pollution Bulletin*, **62**, 651-655.
- Elliott, M., Burdon, D., Hemingway, K.L. & Apitz, S. 2007. Estuarine, coastal and marine ecosystem restoration; confusing management and science- a revision of concepts. *Estuarine, Coastal and Shelf Science*, **74**, 349-366.
- Elliott, M. & Quintino, V. 2007. The Estuarine Quality Paradox, Environmental Homeostasis and the difficulty of detecting anthropogenic stress in naturally stressed areas. *Marine Pollution Bulletin*, **54**, 640-645.
- Elliott, M. & Whitfield, A. 2011. Challenging paradigms in estuarine ecology and management. *Estuarine, Coastal & Shelf Science*, **94**, 306-314.

- Ellis, J.R., Milligan, S.P., Reddy, L., Taylor, N., & Brown, M.J., 2011. *Spawning and nursery grounds of selected fish species in UK waters*. CEFAS Science Series Technical Report no. 147, CEFAS, Lowestoft.
- Elsässer, B., Farinas-Franco, J.M., Wilson, C.D., Kregting, L., & Roberts, D. 2013. Identifying optimal sites for natural recovery and restoration of impacted biogenic habitats in a special area of conservation using hydrodynamic and habitat suitability modelling. *Journal of Sea Research*, **77**, 11-21.
- Engelhard, G.H., van der Kooij, J., Bell, E.D., Pinnegar, J.K., Blanchard, J.L., Mackinson, S. & Righton, D.A. 2008. Fishing mortality versus natural predation on diurnally migrating sandeels *Ammodytes marinus*. *Marine Ecology Progress Series*, **369**, 213-227
- Ewers, R.M. & Didham, R.K. 2006. Confounding factors in the detection of species responses to habitat fragmentation. *Biological Reviews*, **81**, 117-142.
- Ewins, P.I. 1986. *The ecology of Black Guillemots Cepphus grylle in Shetland*. D.Phil. thesis, Oxford. University.
- Ewins, P.J. & D.A. Kirk. 1988. The distribution of Black Guillemots *Cepphus grille* outside the breeding season. *Seabird*, **11**, 50-61.
- Ewins, P.J. 1989. The breeding biology of Black Guillemots *Cepphus grille* in Shetland. *Ibis* **131**, 507-20.
- Ewins, P. J. 1990. The diet of Black Guillemots *Cepphus grylle* in Shetland. *Holarctic Ecology*, **13**, 90-97.
- Fahy, E., Carroll, J. & O'Toole, M. 2003. A preliminary account of fisheries for the surf clam *Spisula solida* (L) (Mactracea) in Ireland. **Fisheries Bulletin No.21**. ISSN 0332-4338.
- Fariñas-Franco, J. M., Allcock, L., Smyth, D., & Roberts, D. 2013. Community convergence and recruitment of keystone species as performance indicators of artificial reefs. *Journal of Sea Research*, **78**, 59-74.
- Fariñas-Franco, J. M. & Roberts, D. 2014. Early faunal successional patterns in artificial reefs used for restoration of impacted biogenic reefs. *Hydrobiologia*, **727**, 75-94.
- Farrow, G. E., Allen, N. H., & Akpan, E. B. (1984). Bioclastic Carbonate Sedimentation on a High-latitude, Tide-dominated Shelf: Northeast Orkney Islands, Scotland (1). *Journal of Sedimentary Research*, **54(2)**, 373-393.
- Foden, J., Rogers, S.I., & Jones, A.P., 2009. Recovery rates of UK seabed habitats after cessation of aggregate extraction. *Marine Ecology Progress Series*, **390**, 15- 26.
- Foster, M. S. 2001. Rhodoliths: Between rocks and soft places. *Journal of Phycology*, **37(5)**, 659-667.
- Frederiksen, M., Wanless, S., Harris, M. P., Rothery, P. & Wilson, L. J. 2004. The role of industrial fisheries and oceanographic change in the decline of North Sea black-legged kittiwakes. *Journal of Applied Ecology*, **41**, 1129-1139.
- Fryganiotis, K., Antoniadou, C. and Chintiroglou, C. 2013. Comparative distribution of the fan mussel *Atrina fragilis* (Bivalva, Pinnidae) in protected and trawled areas of the north Aegean Sea (Thermaikos Gulf). *Mediterranean Marine Science*, **14 (1)**, 119-124.

- Furness, R. W. 2003. Impacts of fisheries on seabird communities. *Scientia Marina*, **67**, 33–45.
- Furness, R.W. & Tasker, M.L. 2000. Seabird-fishery interactions: quantifying the sensitivity of seabirds to reductions in sandeel abundance, and identification of key areas for sensitive seabirds in the North Sea. *Marine Ecology Progress Series*, **202**, 253-264.
- Furness, R. W., Wade, H. M., Robbins, A. M. C. & Masden, E. A. 2012. Assessing the sensitivity of seabird populations to adverse effects from tidal stream turbines and wave energy devices. *ICES Journal of Marine Science*, **69**, 1466-1479.
- Galbraith, R.D., Rice, A. & Strange, E.S. 2004. *An introduction to commercial fishing gear and methods used in Scotland*. Scottish Fisheries Information Pamphlet. No 25.
- Gallego, A., Gibb, F.M., Tullett, D. & Wright, P.J. 2013. Connectivity of Benthic Priority Marine Species within the Scottish MPA Network. *Scottish Marine and Freshwater Science*, **4(2)**, 55pp
- Gardner, M. J., Cottingham, A. Hesp, S. A. Chaplin, J. A. Jenkins, G. I. & Phillips, N. M. 2013. Biological and genetic characteristics of restocked and wild *Acanthopagrus butcheri* (Sparidae) in a southwestern Australian estuary. *Reviews in Fisheries Science*, **21**, 441–453.
- Gaspar, M.B. & Monteiro, C.C. 1999. Gametogenesis and spawning in the subtidal white clam *Spisula solida*, in relation to temperature. *Journal of the Marine Biological Association of the U.K.*, **79**, 753-755.
- Giesen, W. B. J. T., Van Katwijk, M. M., & Den Hartog, C. 1990. Eelgrass condition and turbidity in the Dutch Wadden Sea. *Aquatic Botany*, **37**, 71-85.
- Gilkinson, K.D., Gordon Jr., D.C., MacIsaac, K.G., McKeown, D.L., Kenchington, E.L.R., Bourbonnais, C. & Vass, W.P. 2005. Immediate impacts and recovery trajectories of macrofaunal communities following hydraulic clam dredging on Banquereau, eastern Canada. *ICES Journal of Marine Science*, **62**, 925-947.
- Gilmour, T.H.J. 1967. The defensive adaptations of *Lima hians* (Mollusca, Bivalvia). *Journal of the Marine Biological Association of the UK*, **47(1)**, 209-221).
- González-Correa, J.M., Torquemada, Y.F. & Lizaso, J.L.S. 2008. Long term effect of beach replenishment on natural recovery of shallow *Posidonia oceanica* meadows. *Estuarine Coastal and Shelf Science*, **76**, 834-844.
- Gosling, E. 2003. Bivalve molluscs. *Biology, ecology and culture*. Oxford: Fishing News Books, Blackwell Publishing, 441pp.
- Grall, J., & Hall-Spencer, J. M. 2003. Problems facing maerl conservation in Brittany. *Aquatic Conservation-Marine and Freshwater Ecosystems*, **13**, 55-64.
- Grassle, J.F. & Sanders, H.L. 1973. Life histories and the role of disturbance. *Deep Sea Research*, **20**, 643-659
- Gray, J.S. & Elliott, M. 2009. *Ecology of marine sediments*. From science to management. 2nd ed. Oxford University Press.
- Greathead, C., Demain, D., Dobby, H., Allan, L. & Weetman, A. 2011. Quantitative assessment of the distribution and abundance of the burrowing megafauna and large

epifauna community in the Fladen fishing ground, Northern North Sea. *Scottish Marine and Freshwater Science*, **2(2)**, 26pp.

Greenstreet, S.P.R., Armstrong, E., Mosegaard, H., Hensen, H., Gibb, I.M., Fraser, H.M., Scott, B.E., Holland, G.J. & Sharples, J., 2006. Variation in the abundance of sandeels *Ammodytes marinus* off southeast Scotland: an evaluation of area-closure fisheries management and stock abundance assessment methods. *ICES Journal of Marine Science*, **63**, 1530- 1550.

Greenstreet, S.P.R., Holland G.J., Fraser, H.M., Robertson, M.R., Doyle, K. & Li, R. 2007. Variation in the abundance, distribution, diet and food consumption rates of gadoid predators in the Wee Bankie/Marr Bank region of the northwestern North Sea, consequences for predator body condition, and the impact of closing the area to sandeel fishing. *FRS Collaborative Report*, **09/07**: 125pp.

Greenstreet, S., Fraser, H., Armstrong, E. & Gibb, I., 2010. Monitoring the consequences of the northwestern North Sea sandeel fishery closure. *Scottish Marine and Freshwater Science*, **1(6)**, 1-34

Greenwood, J.G. 2002. Nesting cavity choice by Black Guillemots *Cephus grille*. *Atlantic Seabirds*, **4**, 99-122.

Griffiths, A. M., Sims, D.W., Cotterell, S.P., El Nagar, A., Ellis, J.R., Lynghammar, A., McHugh, M., Neat, F.C., Pade, N.G., Queiroz, N., Serra-Pereira, B., Rapp, T., Wearmouth, V.J., & Genner, M.J., 2010a. Molecular markers reveal spatially segregated cryptic species in a critically endangered fish, the common skate (*Dipturus batis*). *Proceedings of the Royal Society Biological Sciences*, **277**, 1497- 1503.

Griffiths, A.M., Sims, D.W., Johnson, A., Lynghammer, A., McHugh, M., Bakken, T. & Genner, M.J. 2010b. Levels of connectivity between longnose skate (*Dipturus oxyrinchus*) in the Mediterranean Sea and the north-eastern Atlantic Ocean. *Conservation Genetics*, **12(2)**, 577-582

Haegeler, C. W., Schweigert, J. F., 1985. Distribution and Characteristics of Herring Spawning Grounds and Description of Spawning Behavior. *Canadian Journal of Fisheries and Aquatic Sciences*, **42**, 39-55

Haig, J.A., Gillanders, B.M. & Rouse, G.W. 2012. Live fast, die young: the life cycle of the brooding feather star *Aporometra wilsoni* (Echinodermata: Crinoidea). *Invertebrate Biology*, **131**, 235–243.

Hall, S.J. & Harding, J.C. 1997. Physical disturbance and marine benthic communities: the effects of mechanical cockle harvesting on non-target infauna. *Journal of Applied Ecology*, **34 (2)**, 497-517.

Hall-Spencer, J. 2005. Ban on maerl extraction. *Marine Pollution Bulletin*, **50(2)**, 121-121.

Hall-Spencer, J. M., Froggia, D., Atkinson, R. J. A. & Moore, P. G. 1999. The impact of Rapido trawling for scallops, *Pecten jacobaeus* (L.), on the benthos of the Gulf of Venice. *ICES Journal of Marine Science: Journal du Conseil*, **56 (1)**, 111-124.

Hall-Spencer, J.M. & Moore, P.G. 2000a. *Limaria hians* (Mollusca: Limacea): A neglected reef-forming keystone species. *Aquatic Conservation: Marine and Freshwater Ecosystems*, **10**, 267-278.

- Hall-Spencer, J. M., & Moore, P. G. 2000b. Scallop dredging has profound, long-term impacts on maerl habitats. *ICES Journal of Marine Science*, **57**, 1407–1415.
- Hamasaki, K., & Kitada, S. 2013. Catch fluctuation of Kuruma prawn *Penaeus japonicus* in Japan relative to ocean climate variability and a stock enhancement program. *Reviews in Fisheries Science*, **21**, 454–468.
- Hammerstrom, K.K., Kenworthy, W.J., Whitfield, W.J. & Merello, M.F. 2007. Response and recovery dynamics of seagrasses *Thalassia testudinum* and *Syringodium filiforme* and macroalgae in experimental motor vessel disturbances. *Marine Ecology Progress Series*, **345**, 83-92.
- Harding, J.M., King, S.E., Powell, E.N. & Mann, R. 2008. Decadal trends in age structure and recruitment patterns of ocean quahogs *Arctica islandica* from the mid Atlantic Bight in relation to water temperature. *Journal of Shellfish Research*, **27**, 667-690
- Harris, M.P. & Birkhead, T.R. 1985. *Breeding Ecology of the Atlantic Alcidae*, In: The Atlantic Alcidae (Eds D.N. Nettleship & T.R. Birkhead), pp. 155-204. Academic Press, London.
- Harris, M.P. & Wanless, S. 1991. The importance of the lesser sandeel *Ammodytes marinus* in the diet of the shag *Phalacrocorax aristotelis*. *Ornis Scandinavica*, **22**(4), 375-382
- Haskoning UK Ltd. 2006. Investigation into the impact of marine fish farm deposition on maerl beds. *Scottish Natural Heritage Commissioned Report No. 213 (ROAME No. AHLA10020348)*.
- Heath, M.R., Rasmussen, J., Bailey, M.C., Dunn, J., Fraser, J., Gallegro, A., Hay, S.J., Inglis, M. & Robinson, S., 2012. Larval mortality rates and population dynamics of Lesser Sandeel (*Ammodytes marinus*) in the northwestern North Sea. *Journal of Marine Systems*, **93**, 47- 57.
- Herkül, K., Kotta, J. & Pärnoja, M. 2011. Effect of physical disturbance on the soft sediment benthic macrophyte and invertebrate community in the northern Baltic Sea. *Boreal Environmental Research*, **16 (suppl. A)**, 209-219
- Hersteinsson, P. 1999. *Methods to eradicate the American mink (Mustela vison) in Iceland*. In: Workshop on the Control and Eradication of Non-Native Terrestrial Vertebrates. Report to the Council of Europe 1999.
- Heubeck, M. & Richardson, M. G. 1980. Bird mortality following the Esso Bernicia oil spill, Shetland, December 1978. *Scottish Birds*, **11**, 97–108.
- Heubeck, M., Bird, D.R., Harrop, H.R., Harvey, P.V., Mellor, R.M., Suddaby, D., Tasker, M.L., & Uttley, J.D. 1993. *An assessment of the impact of the Braer oil spill on the Shetland breeding population of tysties Cepphus grylle*. Lerwick, SOTEAG, JNCC, RSPB, SNH. (Unpublished report to the Ecological Steering Group on the Shetland Oil Spill).
- Heubeck, M. 2000. Population trends of Kittiwake *Rissa tridactyla*, Black Guillemot *Cepphus grille* and Common Guillemot *Uria aalge* in Shetland, 1978-98. *Atlantic Seabirds (Special Issue)*, **2 (3/4)**, 227-244.
- Hiebenthal, C., Philipp, E.E.R., Eisenhauer, A. & Wahl, M. 2013. Effects of seawater pCO<sub>2</sub> and temperature on shell growth, shell stability, condition and cellular stress of western Baltic Sea *Mytilus edulis* (L.) and *Arctica islandica* (L.). *Marine Biology*, **160**, 2073-2087.

- Holland, G.J., Greenstreet, S.P.R., Gibb, I.M., Fraser, H.M. & Robertson, M.R., 2005. Identifying sandeel *Ammodytes marinus* sediment habitat preferences in the marine environment. *Marine Ecology Progress Series*, **303**, 269- 282.
- Holling, C.S. 1973, Resilience and stability of ecological systems. *Annual Review of Ecology and Systematics*, **4**, 1-23
- Holt, T. J. Rees, E. I. Hawkins, S.J. & Seed, R. 1998. *Biogenic Reefs volume IX. An overview of dynamic and sensitivity characteristics for conservation management of marine SACs*. Scottish Association for Marine Science (UK Marine SACs Project). 170p.
- Hovel, K.A. & Lipcius, R.N. 2001. Habitat fragmentation in a seagrass landscape: patch size and complexity control in Blue crab survival. *Ecology*, **82(7)**, 1814-1829.
- Howson, C. M., Steel. L., Carruthers, M. & Gillham, K. 2012. Identification of Priority Marine Features in Scottish territorial waters. *Scottish Natural Heritage Commissioned Report No. 388*.
- Hughes, D.J., Ansell, A.D. & Atkinson, R.J.A. 1996. Distribution, ecology and life-cycle of *Maxmuellaria lankesteri* (Echiuria: Bonellidae): a review with notes on field identification. *Journal of the Marine Biological Association of the UK*, **76**, 897-908
- Hulleberg, J. 1975. Selective feeding by *Abarenicola pacifica* with notes on *Abarenicola vagabunda* and the concept of gardening in lugworms. *Ophelia*, **14**, 113-137.
- ICES, 2006. *Report of the Working Group on the Assessment of the Demersal Stocks in the North Sea and Skagerrak. Annex 2: Stock Annexes*. 11 Quality handbook: Sandeel in Sub-Area IV. ICES CM 2006/ACFM: 35.
- ICES, 2009. *Report of the ICES Advisory Committee. Section 6.4.22 Sandeel in Division IV excluding the Shetland area*. International Council for the Exploration of the Sea, pp. 214–225.
- ICES. 2012. *Working Group on Methods of Fish Stock Assessments (WGMG)*, 8-12 October 2012, Lisbon, Portugal. ICES CM 2012/SSGSUE: 09. 249 pp.
- Iglesias, S.P., Toulhoat, L. & Sellos, D.Y., 2010. Taxonomic confusion and market mislabelling of threatened skates: important consequences for their conservation status. *Aquatic Conservation: Marine and Freshwater Ecosystems*, **20**, 319- 333.
- Irvine L.M. & Chamberlain Y.M. 1994. *Seaweeds of the British Isles, vol. 1. Rhodophyta Part 2B Corallinales Hildenbrandiales*. pp. vii + 276. London: HMSO.
- Jackson, A. 2007. *Phymatolithon calcareum*. Maerl. Marine Life Information Network: Biology and Sensitivity Key Information Sub-programme [on-line]. Plymouth: Marine Biological Association of the United Kingdom. [cited 11/09/2013]. Available from: <http://www.marlin.ac.uk/speciesfullreview.php?speciesID=4121>.
- Jackson, A. 2008. *Neopentadactyla mixta*. Gravel sea cucumber. Marine Life Information Network: Biology and Sensitivity Key Information Sub-programme [on-line]. Plymouth: Marine Biological Association of the United Kingdom. [cited 16/09/2013]. Available from: <http://www.marlin.ac.uk/speciesfullreview.php?speciesID=3887>.
- Jackson, A. & 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. [cited 11/09/2013]. Available at: <http://www.marlin.ac.uk/speciesfullreview.php?speciesID=3997>.

Jasim, A. K., & Brand, A. R. 1989. Observation on the reproduction of *Modiolus modiolus* in Isle of Man waters. *Journal of the Marine Biological Association of the United Kingdom*, **69(2)**, 373-385.

Jennings, S. & Beverton, R.J.H. 1991. Intraspecific variation in the life history tactics of Atlantic herring (*Clupea harengus* L.) stocks. *ICES Journal of Marine Science*, **48**, 117-125

Jensen, H. & Christensen, A. 2008. RECLAIM. Resolving Climatic Impacts on fish stocks. Specific Targeted Research Project on: Modernisation and sustainability of fisheries, including aquaculture-based production systems. 1.6 Report of WP1. Chapter 18-Sandeel.

Joaquim, S., Gaspar, M.B., Matias, D., Ben-Hamadou, R. & Arnold, W.S. 2008. Rebuilding viable spawner patches of the overfished *Spisula solida* (Mollusca: Bivalvia): a preliminary contribution to fishery sustainability. *ICES Journal of Marine Science*, **65**, 60-64.

Juinio-Menez, M. A., Evangelio, J. C., Olavides, R. D., Pana, M. A. S., de Peralta, G. M. C., Edullantes, M. A., Rodriguez, B. D. R. & Casilagan. I. L.N. 2013. Population dynamics of cultured *Holothuria scabra* in a sea ranch: Implications for stock restoration. *Reviews in Fisheries Sciences*, **21**, 424-432.

Kaiser, M. J., Clarke, K. R., Hinz, H., Austen, M. C. V., Somerfield, P. J. & Karakassis, I. 2006. Global analysis of response and recovery of benthic biota to fishing. *Marine Ecology Progress Series*, **311**, 1-14.

Kelly, J.R. & Harwell, M.A. 1990. Indicators of ecosystem recovery. *Environmental Management*, **14(5)**, 527-545.

Kennedy, R. J., & 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)**, 153-159.

Kennedy, V. S., Breitburg, D.L., Christman, M.C., Luckenbach, M.W., Paynter, K, Kramer, J. & Mann, R. 2011. Lessons learned from efforts to restore oyster populations in Maryland and Virginia, 1990 to 2007. *Journal of Shellfish Research*, **30(3)**, 719-731.

Kenny A. J. & Rees .H.L. 1996. The effects of marine gravel extraction on the macrobenthos: results 2 years post-dredging. *Marine Pollution Bulletin*, **32**, 615-622.

Kennedy, R.J. & 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 (2)**, 79-88.

Korringa, P. 1941. Experiments and observations on swarming, pelagic life and setting in the European flat oyster *Ostrta edulis*. L. *Archives Néerlandaises de Zoologie*, **5**, 1-249.

Kramer, A. M., Dennis, B., Liebhold, A. M., & Drake, J. M. 2009. The evidence for Allee effects. *Population Ecology*, **51(3)**, 341-354.

Krebs, C.J. 1988. *The message of ecology*. Harper & Row, New York, NY

Laing, I., Walker, P., & Areal, F. 2006. Return of the native - is European oyster (*Ostrea edulis*) stock restoration in the UK feasible? *Aquatic Living Resources*, **19(3)**, 283-287.

- Lancaster, J. (Ed.), McCallum, S., Lowe A.C., Taylor, E., Chapman A. 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*.
- Lebata-Ramos, M. J. H., Doyola-Solis, E. F. C. Abroguena, J. B. R. Ogata, H. Sumbing, J. G. & Sibonga, R. C. 2013. Evaluation of post-release behavior, recapture, and growth rates of hatchery-reared abalone *Haliotis asinina* released in Sagay Marine Reserve, Philippines. *Reviews in Fisheries Science*, **21**, 433–440.
- Lei, S A. 2010. Benefits and costs of vegetative and sexual reproduction in perennial plants: a review of literature. *Journal of the Arizona-Nevada Academy of Science*, **42(1)**, 9-14.
- Lindenbaum, C., Bennell, J. D., Rees, E. L. S., McClean, D., Cook, W., Wheeler, A. J., & Sanderson, W. G. 2008. Small-scale variation within a *Modiolus modiolus* (Mollusca: Bivalvia) reef in the Irish Sea: I. Seabed mapping and reef morphology. *Journal of the Marine Biological Association of the United Kingdom*, **88(1)**, 133-141.
- Little, W., 1998. Tope and skate tagging off west Scotland: Part 2. *Glaucus*, **9**, 36–38.
- Lorenzen, K., Agnalt, A., Blankenship, L., Hines, A.J., Leber, K., Loneragan, N.R. & Taylor M.D. 2013. Evolving Context and Maturing Science: Aquaculture-Based Enhancement and Restoration Enter the Marine Fisheries Management Toolbox. *Reviews in Fisheries Science*, **21(3–4)**, 213–221.
- Lotze, H.K., Coll, M., Magera, A.M., Ward-Paige, C. & Airoldi, L. 2011 Recovery of marine animal populations and ecosystems. *Trends in Ecology and Evolution*, **26(11)**, 595-605.
- Lundquist, C.J., Thrush, S.F., Hewitt, J.E., Halliday, J., MacDonald, I. & Cummings, V.J. 2006. Spatial variability in recolonisation potential: influence of organism behaviour and hydrodynamics on the distribution of macrofaunal colonists. *Marine Ecology Progress Series*, **324**, 67-81.
- Lutz, R.A., Mann, R., Goodsell, J.G. & Castanga, M. 1982. Larval and early post-larval development of *Arctica islandica*. *Journal of the marine Biological Association of the UK*, **62**, 745-769
- Macleod, C. K., Crawford, C. M. & Moltschaniwskyj, N. A. 2004. Assessment of long term change in sediment condition after organic enrichment: defining recovery. *Marine Pollution Bulletin*, **49**, 79-88.
- Magorrian, B. H., & Service, M. 1998. Analysis of underwater visual data to identify the impact of physical disturbance on horse mussel (*Modiolus modiolus*) beds. *Marine Pollution Bulletin*, **36(5)**, 354-359.
- Mane, U.H. & Nagabhushanam, R. 1979. Studies on the growth and density of the clam *Paphia laterisulca* at Kalbadevi estuary, Ratnagiri, on the west coast of India. *Malacologia*, **181**, 297-313.
- Malzahn, A.M., Clemmesen C., Wiltshire K.H., Laakmann S. & Boersma M., 2007. Comparative nutritional condition of larval dab *Limanda limanda* and lesser sandeel *Ammodytes marinus* in a highly variable environment. *Marine Ecology Progress Series*. **334**, 205-212.

- Mann, R., & Powell, E.N. 2007. Why oyster restoration goals in the Chesapeake Bay are not and probably cannot be achieved. *Journal of Shellfish Research*, **26(4)**, 905-917.
- Maravelias, C.D., Reid, D.G., & Swartzman, G., 2000. Seabed substrate, water depth and zooplankton as determinants of the pre-spawning spatial aggregation of North Atlantic herring. *Marine Ecology Progress Series*, **195**, 249- 259.
- Marine Ecological Surveys Limited. 2008. *Marine macrofauna genus traits handbook*. Marine Ecological Surveys Limited,. 184pp. ISBN 978-0-9506920-2-9.
- Marine Life Information Network (MarLIN). 2013. Sensitivity Assessment Rationale. [www.marlin.ac.uk/sensitivityrationale.php](http://www.marlin.ac.uk/sensitivityrationale.php). Accessed 2nd August, 2013.
- Marine Scotland, Scottish Government, SNH & JNCC. 2011. Marine Protected Areas in Scotland's Sea: Guidelines on the selection of MPAs and development of the MPA network. Available at <http://www.scotland.gov.uk/Resource/Doc/295194/0114024.pdf>.
- MarLIN, 2006. BIOTIC - Biological Traits Information Catalogue. Marine Life Information Network. Plymouth: Marine Biological Association of the United Kingdom. [Cited insert date] Available from [www.marlin.ac.uk/biotic](http://www.marlin.ac.uk/biotic).
- Mazik, K. & Smyth, K. 2013. *Is 'minimising the footprint' an effective intervention to maximise the recovery of intertidal sediments from disturbance? Phase 1: Literature review*. Natural England Commissioned Reports, Number 110. <http://publications.naturalengland.org.uk/publication/5091106>.
- Mazik, K., Solyanko, K., Brown, S., Mander, L. & Elliott, M. 2010. Role of managed realignment in compensation for habitat loss: a short term solution to a long term problem? *Estuarine Coastal and Shelf Science*, **90**, 11-20.
- McLeod, E., Salm, R., Green, A. & Almany, J. 2004. Designing marine protected area networks to address the impacts of climate change. *Frontiers in Ecology and the Environment*, **7(7)**, 362-370.
- McLeod, E., Salm, R., Green, A. & Almany, J. 2009. Designing marine protected area networks to address the impacts of climate change. *Frontiers in Ecology and the Environment*, **7**, 362-370.
- McLuscky, D.S. and Elliott, M., 2004. *The estuarine ecosystem: ecology, threats, and management*. Oxford: Oxford University Press, page 214.
- Minchin, D. 1995. Recovery of a population of the flame shell, *Lima hians*, in an Irish bay previously contaminated with TBT. *Environmental Pollution*, **90(2)**, 259-262.
- Mitchell, P.I., Newton, S.F., Ratcliffe, N. & Dunn, T.E. 2004. *Seabird populations of Britain and Ireland*. T. & A.D. Poyser, London
- Miyakoshi, Y., Nagata, M. Kitada, S. & Kaeriyama. M. 2013. Historical and current hatchery programs and management of chum salmon in Hokkaido, northern Japan. *Reviews in Fisheries Science*, **21**, 469–479.
- Morrison, J.A., Napier, I.R. & Gamble, J.G. 1991. Mass mortality of herring eggs associated with a sedimenting diatom bloom. *ICES Journal of Marine Science*, **48**, 237-245.

- Morton, R. 2011. The biology and functional morphology of *Arctica islandica* (Bivalvia: Arctiidae): a gerontophilic living fossil. *Marine Biology Research*, **7**, 540-553.
- Mucientes, G.R., Queiroz, N., Sousa, L.L., Tarroso, P., Sims, D.W., 2009. Sexual segregation of pelagic sharks and the potential threat from fisheries. *Biology Letters*, **5**, 156–159.
- Mulcrone, R. 2005. "Holothuroidea" (On-line), Animal Diversity Web. Accessed October 25, 2013 at <http://animaldiversity.ummz.umich.edu/accounts/Holothuroidea/>.
- Navarro, J. M., & Thompson, R. J. 1996. Physiological energetics of the horse mussel *Modiolus modiolus* in a cold ocean environment. *Marine Ecology Progress Series*, **138(1-3)**, 135-148.
- Navarro, J. M., & Thompson, R. J. 1997. Biodeposition by the horse mussel *Modiolus modiolus* (Dillwyn) during the spring diatom bloom. *Journal of Experimental Marine Biology and Ecology*, **209(1-2)**, 1-13.
- Neckles, H.A., Short, F.T., Barker, S. & Kopp, B.S. 2005. Disturbance of eelgrass *Zostera marina* by commercial mussel *Mytilus edulis* harvesting. *Marine Ecology Progress Series*, **285**, 57-73.
- Neal, K., Pizzolla, P. & Wilding, C. 2008. *Dipturus batis*. Common Skate. Marine Life Information Network: Biology and Sensitivity Key Information Sub-programme [on-line]. Plymouth: Marine Biological Association of the United Kingdom. Available from: <http://www.marlin.ac.uk/speciesbenchmarks.php?speciesID=3183>. Accessed on 25/08/2013
- Newell, R. C., Seiderer, L.J. & 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**, 127-178.
- Nichols, D. 1994. Reproductive Seasonality in the Comatulid Crinoid *Antedon bifida* (Pennant) from the English Channel. *Philosophical Transactions: Biological Sciences*, **343** (1034), 113-134.
- Nichols, D. 1991. *Seasonal reproductive periodicity in the European comatulid crinoid Antedon bifida* (Pennant). In Biology of Echinodermata. Proceedings of the Seventh International Echinoderm Conference, Atami, 9-14 September 1990 (ed. T. Yanagisawa, I. Yasumasu, C. Oguro, N. Suzuki & T. Motokawa), 241-248.
- Nordström, M., Högmander, J., Laine, J., Nummelin, J., Laanetu, N. & Korpimäki, E. 2003. Effects of feral mink removal on seabirds, waders and passerines on small islands in the Baltic Sea. *Biological Conservation*, **109**, 359-368.
- Norton T. A. 1992. Dispersal by macroalgae. *British Phycological Journal*. **27**, 293-301.
- O'Connor, B. 1981. Some Echinoderms from the west coast new to the Irish Fauna. *The Irish Naturalists Journal*, **20 (6)**, 247-249.
- Okill, D. 2002. *Black Guillemot*. In Wernham, C.V., Toms, M.P., Marchant, J.H., Clark, J.A. Siriwardena, G.M. & Baillie, S.R. (eds.) The Migration Atlas: Movements of the Birds of Britain and Ireland: 405–406. T. & A.D. Poyser, London.

- O'Neill, F.G., Robertson, M., Summerbell, K., Breen, M. & Robinson, L.A. 2013. The mobilisation of sediment and benthic infauna by scallop dredges. *Marine Environmental Research*, **90**, 104-112.
- Oulasvirta, P., & Lehtonen, H., 1988. Effects of sand extraction on herring spawning and fishing in the Gulf of Finland. *Marine Pollution Bulletin*, **8**, 383- 386.
- Parrish, B.B., & Saville A., 1965. The biology of the Northeast Atlantic Herring populations. *Oceanography Marine Biology: An Annual Review*, **3**, 323-373.
- Pawson, D. L. 1982. *Holothuroidea*. In: Parker, S. P., ed. *Synopsis and Classification of Living Organisms*. New York: McGraw-Hill, 813-818.
- Pechenik J. A. 1999. On the advantages and disadvantages of larval stages in benthic marine invertebrate life cycles. *Marine Ecology Progress Series*, **177**, 269-297.
- Powell, EN. & Mann, R. 2005. Evidence of recruitment in the ocean quahog *Arctica islandica* in the mid-Atlantic Bight. *Journal of Shellfish Research*, **24**, 517-530
- Ragnarsson, S. A., & Burgos, J. M. 2012. Separating the effects of a habitat modifier, *Modiolus modiolus* and substrate properties on the associated megafauna. *Journal of Sea Research*, **72**, 55-63.
- Rees, E. I. S., Sanderson, W. G., Mackie, A. S. Y., & Holt, R. H. F. 2008. Small-scale variation within a *Modiolus modiolus* (Mollusca: Bivalvia) reef in the Irish Sea. III. Crevice, sediment infauna and epifauna from targeted cores. *Journal of the Marine Biological Association of the United Kingdom*, **88(1)**, 151-156.
- Rees I. 2009. *Assessment of Modiolus modiolus Beds in the OSPAR Area*. OSPAR Background Document for *Modiolus modiolus*. To the OSPAR Commission, on behalf of Joint Nature Conservation (JNCC), Peterborough. 22 pages.
- Richardson, C. A., Kennedy, H. Duarte, C. M. Kennedy, D. P. & Proud, S. V. 1999. Age and growth of the fan mussel *Pinna nobilis* from south-east Spanish Mediterranean seagrass (*Posidonia oceanica*) meadows. *Marine Biology*, **133 (2)**, 205-212.
- Ridgway, I.D., Richardson, C.A., Scourse, J.D., Butler, P.G. & Reynolds, D. J. 2012. The population structure and biology of the ocean quahog, *Arctica islandica*, in Belfast Lough, Northern Ireland. *Journal of the Marine Biological Association of the United Kingdom*, **92(3)**, 539-546
- Roberts, C. D. 1975. Investigations into a *Modiolus modiolus* (L.) (Mollusca: Bivalvia) community in Strangford Lough, N. Ireland. Report Underwater Association New Series, **1**, 27-49.
- Roberts, D. Allcock, L., Fariñas-Franco, J.M., Gorman, E., Maggs, C., Mahon, A.M., Smyth, D., Strain, E. & Wilson, C.D. 2011. *Modiolus restoration research project. Final report and recommendations*. Report to Department of Agriculture and Rural Development and the Northern Ireland Environment Agency by Queens University, Belfast. 246pp.
- Roberts, C.M., Branch, G., Bustamante, R.H, Castilla, J.C., Dugan, J., Halpern, B.J., Lafferty, K.D., Leslie, H., Lubchenco, J., McArdle, D., Ruckelhaus, M. & Warner, R.R. 2003. Application of ecological criteria in selecting marine reserves and developing reserve networks. *Ecological Applications*, **13(1)**, S215-S228

Roberts, D., Davies, C., Mitchell, A., Moore, H., Picton, B., Portig, A., Preston, J., Service, M., Smyth, D., Strong, D. and Vize, S. 2004. *Strangford Lough Ecological Change Investigation (SLECI)*. Report to Environment and Heritage Service by the Queen's University, Belfast.

Roberts, D., Smyth, D., & Browne, L. 2005. Pros and cons of small scale native oyster restoration programmes: Experiences gained in Strangford Lough, Northern Ireland. *Journal of Shellfish Research*, **24**(1), 335-335.

Ronconi, R. A. & Clair, C. C. S. 2002. Management options to reduce boat disturbance on foraging black guillemots (*Cephus grylle*) in the Bay of Fundy. *Biological Conservation*, **108**, 265-271.

Ropes, J.W. 1985. Modern methods to age oceanic bivalves. *Nautilus*, **99**, 53-57.

Rosenberg, A., Bigford, T. E., Leathery, S., Hill, R. L., & Bickers, K. 2000. Ecosystem approaches to fishery management through essential fish habitat. *Bulletin of Marine Science*, **66**(3), 535-542.

Rothschild, B. J., Ault, J. S., Gouletquer, P., & Heral, M. 1994. Decline of the Chesapeake Bay oyster population – a century of habitat destruction and overfishing. *Marine Ecology Progress Series*, **111**(1-2), 29-39.

Rumohr H., & Krost, P. 1991. Experimental evidence of damage to benthos by bottom trawling with special reference to *Arctica islandica*. *Meeresforschung/Reports on Marine Research*, **33**, 340-45.

Rowley, S. and Wilding, C. 2008. *Ammodytes tobianus*. Lesser sand eel. Marine Life Information Network: Biology and Sensitivity Key Information Sub-programme [on-line]. Plymouth: Marine Biological Association of the United Kingdom. [cited 21/10/2013]. Available from: <http://www.marlin.ac.uk/speciesinformation.php?speciesID=2480>.

Sanderson, W. G., Holt, R. H. F., Kay, L., Ramsay, K., Perrins, J., McMath, A. J., & Rees, E. I. S. 2008. Small-scale variation within a *Modiolus modiolus* (Mollusca: Bivalvia) reef in the Irish Sea. II. Epifauna recorded by divers and cameras. *Journal of the Marine Biological Association of the United Kingdom*, **88**(1), 143-149.

Santos, S.L. & Simon, J.L. 1980. Marine soft-bottom community establishment following annual defaunation: larval or adult recruitment. *Marine Ecology Progress Series*, **2**, 235-241.

Saville, A. & R.S. Bailey. 1980. The assessment and management of the herring stocks in the North Sea and to the west of Scotland. *Rapports et Procès-Verbaux des Réunions du Conseil Permanent International pour l'Exploration de la Mer*, **177**, 112-142.

Scheffer, M., Bascompte, J., Brock, W.A., Brovkin, V. Carpenter, S.R., Dakos, V., Held, H., van Nes, E.H., Rietkerk, M. & Sugihara, G. 2009. Early-warning signals for critical transitions. *Nature*, **461**, 53–59.

Schmidt, J.O., Van Damme, C.J.G., Rockmann, C., & Dickey- Collas, M., 2009. Recolonisation of spawning grounds in a recovering fish stock: recent changes in North Sea herring. *Scientia Marina*, **73**S1, 153- 157.

Schratzberger, M. & Warwick, R.M. 1999. Differential effects of various types of disturbances on the structure of nematode assemblages: an experimental approach. *Marine Ecology Progress Series*, **181**, 227-236.

Schulte, David M., Burke, Russell P., & Lipcius, Romuald N. 2009. Unprecedented Restoration of a Native Oyster Metapopulation. *Science*, **325(5944)**, 1124-1128.

Scottish Government. 2013. FEAST – Feature Activity Sensitivity Tool. <http://www.marine.scotland.gov.uk/FEAST/Index.aspx>. Accessed 20/10/2013

Scottish Natural Heritage (SNH) and the Joint Nature Conservation Committee (JNCC). 2013. Advice to the Scottish Government on the selection of Nature Conservation Marine Protected Areas (MPAs) for the development of the Scottish MPA network. *Scottish Natural Heritage Commissioned Report No. 547*.

Scottish Natural Heritage. 2013. *Flame shell beds*. <http://www.snh.gov.uk/about-scotlands-nature/species/invertebrates/marine-invertebrates/flame-shell-beds/>. Accessed 09/08/2013.

Scottish Natural Heritage. 2014. *Site summary Loch Sween Marine Protected Area*. <http://www.snh.gov.uk/docs/A978504.pdf>. Accessed 29/09/2014

Seaman, W. 2007. Artificial habitats and the restoration of degraded marine ecosystems and fisheries. *Hydrobiologia*, **580**, 143-155.

Seed, R., & Brown, R. A. 1975. The influence of reproductive cycle, growth, and mortality on population structure in *Modiolus modiolus* (L.) *Cerastoderma edule* (L.) and *Mytilus edulis* L., (Mollusca: Bivalvia) in: Barnes, H.B. (Ed.). *Ninth European Marine Biology Symposium*. pp. 257-274.

Seed, R., & Brown, R. A. 1978. Growth as a strategy for survival in 2 marine bivalves, *Cerastoderma edule* and *Modiolus modiolus*. *Journal of Animal Ecology*, **47(1)**, 283-292.

Shelmerdine, R. L., & Leslie, B. 2009. Restocking of the native oyster, *Ostrea edulis*, in Shetland: habitat identification study. *Scottish Natural Heritage Commissioned Report No. 396*, i-iv, 1-26.

Shimeta, J. & Jumars, P. A. 1991. Physical-mechanisms and rates of particle capture by suspension-feeders. *Oceanography and Marine Biology*, **29**, 191-257.

Smith, W.G., & Morse, W.W., 1993. Larval distribution patterns: early signals for the collapse/ recovery of Atlantic herring *Clupea harengus* in the Georges Bank area. *Fishery Bulletin*, **91**, 338- 347.

Smyth, D., Roberts, D., & Browne, L. 2009. Impacts of unregulated harvesting on a recovering stock of native oysters (*Ostrea edulis*). *Marine Pollution Bulletin*, **58(6)**, 916-922.

Snelgrove, P.V.R. & Butman, C.A. 1994. Animal-sediment relationships revisited: cause versus affect. *Oceanography and Marine Biology: Annual Review*, **32**,111-177.

SNH document B988821. Maerl beds. <http://www.snh.gov.uk/docs/B988821.pdf>.

Solandt, J.-L. 2003. *Atrina fragilis* (Pennant 1777): A Species of Conservation Concern. *British Wildlife*, **14 (6)**, 423-427.

Steller, D. L., Riosmena-Rodriguez, R., Foster, M. S., & Roberts, C. A. 2003. Rhodolith bed diversity in the Gulf of California: the importance of rhodolith structure and consequences of disturbance. *Aquatic Conservation-Marine and Freshwater Ecosystems*, **13**, 5-20.

Tasker, M. L., Camphuysen, C. J., Cooper, J., Garthe, S., Montevecchi, W. A. & Blaber, S. J. M. 2000. The impacts of fishing on marine birds. *ICES Journal of Marine Science*, **57**, 531-547.

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. & Wilkinson, M. (2013). Framework for understanding marine ecosystem health. *Marine Ecology Progress Series*, **494**, 1–27 + suppl. material.

Thompson, T., Jones, D. & Dreibelbis, D. 1980. Annual internal growth banding and life history of the Ocean quahog *Arctica islandica* (Mollusca: Bivalvia). *Marine Biology*, **57**, 25 - 34.

Thrush, S.F., Halliday, J., Hewitt, J.E. & Lohrer, A. 2008. The effects of habitat loss, fragmentation and community homogenization on resilience in estuaries. *Ecological Applications*, **18**(1), 12-21.

Thrush, S.F., Hewitt, J.E., Norkko, A., Cummings, V.J. Funnell, G.A. 2003. Macrobenthic recovery processes following catastrophic sedimentation on estuarine sandflats. *Ecological Applications*, **13** (5), 1433-1455

Thrush, S.F., Whitlatch, R.B., Pridmore, R.D., Hewitt, J.E., Cummings, V.J. & Wilkinson, M.R. 1996. Scale dependent recolonization: the role of sediment stability in a dynamic sandflat habitat. *Ecology*, **77**(8), 2472-2487.

Trigg, C. 2009. Ecological studies on the bivalve *Limaria hians* (Gmelin). Unpublished PhD thesis, Heriot-Watt University.

Trigg, C. & Moore, C.G. 2009. Recovery of the biogenic nest habitat of *Limaria hians* (Mollusca: Limacea) following anthropogenic disturbance. *Estuarine, Coastal & Shelf Science*, **82**, 351-356.

Torrents, O., Garrabou, J., Marschal, C. & Harmelin, J.G. 2005. Age and size at first reproduction in the commercially exploited red coral *Corallium rubrum* (L.) in the Marseilles area (France, NW Mediterranean). *Biological Conservation*, **121**, 391–397.

Tyler-Walters, H. 2003. *Limaria hians* beds in tide-swept sublittoral muddy mixed sediment. Marine Life Information Network: Biology and Sensitivity Key Information Sub-programme [on-line]. Plymouth Marine Biological Association of the United Kingdom. Accessed on 07/08/2013. Available from: <http://www.marlin.ac.uk/habitatsensitivity.php?habitatid=112&code=1997>.

Tyler-Walters, H. 2007. *Modiolus modiolus*. Horse mussel. Marine Life Information Network: Biology and Sensitivity Key Information Sub-programme [on-line]. Plymouth: Marine Biological Association of the United Kingdom. Available from: <http://www.marlin.ac.uk/speciesfullreview.php?speciesID=3817>.

Tyler-Walters, H 2008. *Limaria hians*. Gaping file shell. Marine Life Information Network: Biology and Sensitivity Key Information Sub-programme [on-line]. Plymouth Marine Biological Association of the United Kingdom. Accessed on 07/08/2013. Available from: <http://www.marlin.ac.uk/speciesfullreview.php?speciesID=3679>.

Tyler-Walters, 2011. Slipper limpet, *Crepidula fornicata*. <https://secure.fera.defra.gov.uk/nonnativespecies/factsheet/downloadFactsheet.cfm?speciesId=1028>. Accessed 16/12/13.

Tyler-Walters, H. & Arnold, C. 2008. Sensitivity of intertidal benthic habitats to impacts caused by access to fishing grounds. Report to Countryside Council for Wales from the Marine Life Information Network (MarLIN). Marine Biological Association of the UK, Plymouth. Contract No. FC73-03-327, 48pp.

UK Biodiversity Group. 1999. *Maritime Species and Habitats. Atrina fragilis (a fan shell). Tranche 2 Action Plans*. English Nature, Peterborough, 63-35.

University Marine Biological Station Millport. 2007. Conservation of the Native Oyster *Ostrea edulis* in Scotland. *Scottish Natural Heritage Commissioned Report No. 251 (ROAME No. F02AA408)*.

Valdemarsen, T., Wendelboe, K., Egelund, J.T., Kristensen, E. & Flindt, M.R. 2011. Burial of seeds and seedlings by the lugworm *Arenicola marina* hampers eelgrass (*Zostera marina*) recovery. *Journal of Experimental Marine. Biology and Ecology*, **410**, 45-52.

Van der Kooij, J., Scott, B.E. & Mackinson, S. 2008. The effects of environmental factors on daytime sandeel distribution and abundance on the Dogger Bank. *Journal of Sea Research*, **60**, 3, 201-209.

Van Keeken, O.A., van Hoppe, M., Griff, R.E. & Rijnsdorp, A.D. 2007. Changes in the spatial distribution of North Sea plaice (*Pleuronectes platessa*) and implications for fisheries management. *Journal of Sea Research*, 57(2-3), 187–197.

Vasconcelos, P., Morgado-André, A., Morgado-André, C. & Gaspar, M.B. 2011. Shell strength and fishing damage to the smooth clam (*Callista chione*); simulating impacts caused by bivalve dredging. *ICES Journal of Marine Science*, **68(1)**, 32-42.

Volkenborn, N., Robertson, D.M. & Reise, K. 2009. Sediment destabilizing and stabilizing bio-engineers on tidal flats: cascading effects of experimental exclusion. *Helgoland Marine Research*, **63 (1)**, 27-35.

Walker, P. A. & Hislop, J. R. G. 1998. Sensitive skates or resilient rays? Spatial and temporal shifts in ray species composition in the central and north-western North Sea between 1930 and the present day. *ICES Journal of Marine Science*, **55**, 392–402.

Wanless, S., Frederiksen, M., Daunt, F., Scott, B. E., & Harris, M. P. 2007. Black-legged kittiwakes as indicators of environmental change in the North Sea: Evidence from long-term studies. *Progress in Oceanography*, **72**, 30-38.

Warwick, R. M., McEvoy, A. J. & Thrush, S. F. 1997. The influence of *Atrina zelandica* Gray on meiobenthic nematode diversity and community structure. *Journal of Experimental Marine Biology and Ecology*, **214 (1-2)**, 231-247.

Wearmouth, V.J. & Sims, D.W. 2009. Movement and behaviour patterns of the critically endangered common skate *Dipturus batis* revealed by electronic tagging. *Journal of Experimental Marine Biology and Ecology*, **380**, 77-87.

Wilding, C. & Wilson, E. 2009. *Swiftia pallida*. Northern sea fan. Marine Life Information Network: Biology and Sensitivity Key Information Sub-programme [on-line]. Plymouth:

Marine Biological Association of the United Kingdom. [cited 12/09/2013]. Available from: <http://www.marlin.ac.uk/reproduction.php?speciesID=4407>.

Williams, A., Althaus, F., Dunstan, P.K., Poore, G.C.B., Bax, N.J., Kloser, R.J. & McEnnulty, F.R. 2010. Scales of habitat heterogeneity and megabenthos biodiversity on an extensive Australian continental margin (100-1100 m depths). *Marine Ecology: an evolutionary perspective*, **31**, 222-236.

Wilson, J.G. 1996. Long-term changes in density, population structure and growth rate of *Tellina tenuis* from Dublin Bay, Ireland. *Oceanologica Acta*, **20**, 267-274.

Witbaard, R. & Bergman, M.J.N. 2003. The distribution and population structure of the bivalve *Arctica islandica* L. in the North Sea: what possible factors are involved? *Journal of Sea Research*, **50**, 11-25.

Witbaard, R., Duineveld, G.C.A. & de Wilde, P.A.W.J. 1999. Geographical differences in growth rates of *Arctica islandica* (Mollusca: Bivalvia) from the North Sea and adjacent waters. *Journal of the Marine Biological Association of the UK*, **79**, 907-915.

Woodin, S.A., Lindsay, S.M. & Wethey, D.S. 1995. Process-specific recruitment cues in marine sedimentary systems. *Biological Bulletin*, **189** (1), 49-58.

Woolmer, A.P., Syvret, M. & FitzGerald A. 2011. *Restoration of Native Oyster, Ostrea edulis, in South Wales: Options and Approaches*. CCW Contract Science Report No: 960, 93 pp.

Wright, P.J., Jensen, H. & Tuck, L., 2000. The influence of sediment type on the distribution of the lesser sandeel, *Ammodytes marinus*. *Journal of Sea Research*, **44**, 243- 256.

Wu, R.S.S. & Shin, P.K.S. 1998. Transplant experiments on growth and mortality of the fan mussel *Pinna bicolor*. *Aquaculture*, **163**, 47-62.

Young, C. M. & Thompson, T. E. 1976 *Living Marine Molluscs*. William Collins and Sons and Co.

Zajac, R.N. 2004. Macrofaunal responses to pit-mound patch dynamics in an intertidal mudflat: local versus patch-type effects. *Journal of Experimental Marine Biology Ecology*, **313**, 297-315.

Zajac, R.M. & Whitlatch, R.B. 1982. Responses of estuarine infauna to disturbance I. Spatial and temporal variation of initial recolonization. *Marine Ecology Progress Series*, **10**, 1-14.

Zajac, R.N. & Whitlatch, R.B. 2003. Community and population-level responses to disturbance in a sandflat community. *Journal of Experimental Marine Biology and Ecology*, **294**, 101-125.

Zonfrillo, B & Nogales, M. 1992. First breeding records of Shelduck & Black Guillemot on Ailsa Craig. *Glasgow Naturalist*. **22**(2), 197-198.

## **ANNEX 1: CONSERVATION OBJECTIVES**

Source: Scottish Government Draft Marine Protected Areas Handbook  
<http://www.gov.scot/Topics/marine/marine-environment/mpanetwork/engagement/ManagementHandbook>

A1.1 Conservation objectives for NC MPAs are developed to reflect the purpose defined in the Marine Acts as being for the conservation of marine flora or fauna, or for the conservation of marine habitats or geological / geomorphological features. Conservation objectives form part of the designation order for NC MPAs and will therefore be in place at the time that a site is formally designated.

A1.2 It is important to note that management measures may be required irrespective of the conservation objective. Using a risk-based approach to the management of NC MPAs, measures will be developed and implemented when it is identified they will be necessary to achieve the conservation objectives.

A1.2.1 Conservation objectives for NC MPAs will normally be to “conserve” the feature in the first instance, acknowledging where there is uncertainty regarding feature condition. This does not preclude a “recover” conservation objective being defined at this stage if required by the scientific evidence. The conservation objectives will highlight the qualities of the protected features that need to be covered and link to supporting processes.

A1.2.2 Where evidence exists that a protected feature of an MPA is in good condition, or where limited evidence exists and therefore there is uncertainty about the condition of protected features, then the conservation objective will be to “conserve”. In cases where uncertainty exists this will be recorded and reviewed as further knowledge and evidence emerges.

A1.2.3 Where evidence exists that a protected feature in a NC MPA is declining, and/or has been modified, then the conservation objective will be to “recover” the feature. This assessment will take account of our understanding of natural change.

A1.3 The longer-term aim is to develop SMART (Specific, Measurable, Attainable, Relevant and Timely) conservation objectives for every NC MPA. As a consequence of site monitoring there may also be a need to review the conservation objectives. It is envisaged that when such a review takes place then the objectives will be revised. If there is sufficient information to make the conservation objectives SMART then this will be done at the same time.

## ANNEX 2: SEXUAL MATURITY AND LONGEVITY OF SPECIES OF INTEREST IN SCOTTISH WATERS

Species	Age at sexual maturity	Reference	Longevity	Reference
<b>Bivalves</b>				
<i>Modiolus modiolus</i>	Earliest, 3 years, majority by 5-6 years, latest 7-8 years	Jasim & Brand (1989)	>20 years, habitat dependent	Comely (1978)
<i>Ostrea edulis</i>	3 <sup>rd</sup> year	Jackson & Wilding (2009)	10-15 years, most individuals in a population 2-6 years old	Jackson and Wilding (2009)
<i>Limaria hians</i>	2 <sup>nd</sup> year	Minchin (1995)	-	-
<i>Arctica islandica</i>	4-14 years	Thompson <i>et al.</i> (1980)	200 years	Ropes (1985)
			>400 years	Ridgway <i>et al.</i> (2012)
<b>Components of Shallow tide-swept coarse sands with burrowing bivalves</b>				
<i>Chlamys varia</i>	1- 2 years	Marine Ecological Surveys Limited (2008)	Up to 7	Conan and Schafee (1978)
<i>Moerella</i> spp. ( <i>Tellina</i> )	3	Wilson (1996)	2- 10 years	Marine Ecological Surveys Limited (2008)
<i>Dosinia lupinus</i>	1-2 years	Marine Ecological Surveys Limited (2008)	>10 years	Marine Ecological Surveys Limited (2008)
<i>Timoclea ovata</i>	1 year	Marine Ecological Surveys Limited (2008)	4-6 years	Marine Ecological Surveys Limited (2008)
<i>Chamelea gallina</i>	<1 year	Marine Ecological Surveys Limited (2008)	11- 20 years	Marine Ecological Surveys Limited (2008)
<i>Paphia</i> spp.	-	-	3	Mane and Nagabhusanam (1979)
<i>Ensis</i> spp	1 <sup>st</sup> year	Marine Ecological Surveys Limited (2008)	10- 20 years	Marine Ecological Surveys Limited (2008)
<i>Spisula solida</i>	Within 1 <sup>st</sup> year	Gaspar and Monteiro, 1999	05-Oct	Fahy <i>et al.</i> (2003).
<i>Spatangus</i> spp.	3- 5 years	MarLIN (2006)	11- 20 years	MarLIN (2006)
<i>Antedonidae</i> (family level)	1-2 years depending on species	Thomson (1865, In: Nichols, 1994)	Unkown, but many crinoids reach maturity after 1 year and may remain reproductively active for several years	Haig <i>et al.</i> (2012)

Species	Age at sexual maturity	Reference	Longevity	Reference
<b>Maerl or coarse shell gravel with burrowing cucumbers</b>				
<i>Holothruoidea</i>	-		5 – 10 years	Barnes (1987, In: Mulcrone, 2005)
<i>Echinoderms</i>	1-5 years depending on species	Marine Ecological Surveys Limited (2008)	1-20 years depending on species	Marine Ecological Surveys Limited (2008)
<b>Components of burrowed mud</b>				
<i>Pachycerianthus multiplicatus</i>	unknown		unknown, but thought to be several years	(Hughes, 1998)
<i>Cnidaria</i>	8 weeks- 2 years depending on species	Marine Ecological Surveys Limited (2008)	5 months- 20 years depending on species	Marine Ecological Surveys Limited (2008)
<i>Anthozoa</i>	7- 10 years, reproduction output is higher in larger colonies	Torrents <i>et al.</i> (2005)	-	-
<i>Hydrozoa</i>	-	-	-	-
<i>Virgularia mirabilis</i>	Insufficient info (but similar spp. <i>Ptilosarcus guerneyi</i> 5 or 6 years to reach sexual maturity.	Birkeland, (1974)	Insufficient info, but <i>Ptilosarcus guerneyi</i> up to 15 years	Birkeland (1974)
<i>Funiculina quadrangularis</i> - (Similar info available as for <i>V. mirabilis</i> )	Similar info available as <i>V. mirabilis</i>		Similar info available as <i>V. mirabilis</i>	
<b>Northern sea fan and sponge communities</b>				
<i>Caryophyllia smithii</i>		-	11- 20 years	MarLIN (2006)
<i>Swiftia pallida</i>	No information	-	Up to 20 years	Wilding and Wilson (2009)
<b>Fish</b>				
Sandeels	4-7 years	Rowley and Wilding (2008)		
Common skate	20-100 years	Neal <i>et al.</i> (2008)	11 years	Neal <i>et al.</i> , 2008
Herring	12-20 years	Jennings and Beverton (1991)	3-4 years	US Department of Commerce (1999)

## **ANNEX 3: WORKSHOP TO REVIEW AND REFINE THE FRAMEWORK FOR ASSESSING RECOVERY POTENTIAL**

LOCATION AND DATE: **SNH, PERTH. 7TH FEBRUARY, 2014**

### **Participants**

<b>Name</b>	<b>Organisation</b>
Krysia Mazik	IECS
James Strong	IECS
Mike Elliott	IECS
Steve Barnard	IECS
Sarah Cunningham	SNH
Katie Gilham	SNH
Ben James	SNH
Jane Dodd	SNH
Cathy Tilbrook	SNH
Peter Chanotis	JNCC
David Vaughan	JNCC

### **WORKSHOP SUMMARY**

#### **Recovery definitions**

There was general agreement with the definitions proposed in the report. However, most participants expressed that the format of the presentation had provided greater clarity and it was requested that the format of this section was amended to achieve this level of clarity. In terms of recovery potential it was acknowledged noted that any assessment of recovery must have clearly defined and agreed recovery endpoints. A component of this should be spatial scale and the natural state of receiving/host habitats (e.g. state of adult oyster beds may affect the potential for spat settlement). It was agreed that the recovery endpoint should lie within a range to account for natural variability within both the reference and the recovering population, community or habitat. In discussing the problem of 'convergence' of naturally varying populations of recovering and reference communities potentially giving a (false) indication of the achievement of a recovered state it was agreed that LONG TERM recovery should be clearly defined as part of the overall definition of recovery (e.g. including long term population means or trajectory means as recovery criteria).

It was also noted that species with no documented recovery potential tended to be long-lived, with low fecundity and low dispersal ability. For some species such as maerl, recovery of the associated community must also be considered (e.g. the functional provisions associated with the physical habitat of the maerl bed) and that even if expansion of maerl may not be possible, recovery of the associated species within the existing maerl habitat may be achievable.

It was agreed that trajectory of recovery was as important as achieving a (defined) recovered endpoint as with some species the latter may not be attainable in practical terms due to long timescales.

In setting a recovery target SNH acknowledged that 'pristine condition' as a reference or target is aspirational and not really achievable. Consequently there is a need to move away from the use of pristine conditions and develop more realistic target definitions. Furthermore, in defining a 'Recovered Reference Range', SNH were fully supportive of the pragmatic approach of including measures pertaining to density, spatial extent, and sustainability or self-sufficiency within a Recovered Reference Range definition. Given that the recovery

trajectory also provides an indicator of recovery, consideration should be given to accounting for natural variability or fluctuation around the recovery trajectory (i.e. retain the focus on medium/longer-term trends rather than purely short-term).

For recovering species or communities, the target should strictly be for recruitment to exceed loss, 'holding the line' (where recruitment balances loss) may be the best that can be expected in some situations. The ideal of recovering to a predetermined species composition should be balanced against the understanding that species composition is likely to change through time (as different species recover at different rates) – a dynamic that might lead to inter-specific competitive effects which could alter the ultimate 'climax community'. Perhaps it is better to look for 'typicality', including relative abundance information, rather than having a fixed (absolute) abundance or density targets for a species or community.

For biogenic habitats, it will be important to maintain the notion of bioengineering – species recovery is itself important, but changes to the associated wider community is perhaps equally important as regards the definition of a recovery end point. It was noted that water quality within the water column may also be an important factor in some instances (e.g. through turbidity from suspended solids affecting primary production rates).

Any long-term underlying changes (e.g. due to climate change effects) which might affect the baseline or reference point will need to be considered as part of target definitions. Such external factors may necessitate a change in the reference point or a change in the management of anthropogenic impacts to account for increased vulnerability/sensitivity. In situations where populations might have been affected by such unmanaged exogenous pressures, there is a requirement to ensure that populations are not tipped 'over the edge'. In defining physical habitat recovery points in terms of the long-term similarity of habitat attributes to a pre-defined reference, it is likely that such targets would be better expressed as a range (to encompass natural variability) rather than defined as single target values. Finally, it was noted SNH work suggests that skate tend use deeper water channels and so there may be some merit in moving to protect all habitats within MPAs below a depth of 100 m.

Given the longevity of many species, it is unlikely that many will achieve 'recovered' status within the lifetime of the Scottish MPA project in its current form – this underpins the importance of addressing/halting current declines as well as the value of monitoring the trajectory of recovery to identify a 'favourable change over time'.

### **Framework for assessing recovery potential**

1. **MANAGEMENT OPTIONS** – amend Table 4 to account for seasonality in pressure and/or management response. Account for seasonality and intensity and consider other management measures (e.g. changing the nature of the activity to use less damaging methods) – can the pressure be amended rather than removed.
2. **COST EFFECTIVENESS** – SNH have a responsibility to determine what the likely success of restoration efforts might be. Cost effectiveness needs to be considered in biological terms rather than simply in socio-economic terms. For example, is restoring a large adult population cost effective in the long term even if initially expensive? How should cost effectiveness be measured? E.g. what is the value of a fish nursery ground?

### *Table format*

SNH suggested that it should be presented as a series of questions indicating what the problem is and how it could be solved:

- What is the problem?
  - What needs to be achieved – highlight objectives/definitions of recovery?
  - What are the site/species specific issues and options for management?
1. constraints to recovery – what are the habitat requirements of the feature and do the environmental conditions match these? Is the habitat broadly suitable? What are the biological constraints (species life history traits)?
  2. What are the management options – pressure removal is just one option

Comments from SNH on way forward:

- Can we use a framework to record 'soft intelligence' - e.g. a tick list to provide a 'probability of evidence' or 'weight of evidence' type approach to identify/define recovery endpoints (or to demonstrate recovery towards and endpoint)?
- Can the language be simplified & clarified (to facilitate ready sharing with stakeholders) - as the information is presented in the PowerPoint presentation it appears a bit wordy and hard to follow.
- Is it possible to provide examples of 'how' things would be done (e.g. where recruitment exceeds loss you would expect an increase in the number of individuals in a given area ... so we can test or monitor against this by undertaking quadrat counts, which should show upward trend through time ...)

The first stage (what information do we already have) should be presented as a set of prompts. (what should we be looking for and do we have it, in terms of habitat requirements). We need to consider what can be done with the existing habitat. This needs to be emphasised in the supporting text. The text should also indicate that these points are of equal importance.

Key steps are identifying the problem, identifying the cause, identifying the solution. Clarification of managing the cause and the consequences of decline is required. Managing the consequences refers to the knock on effects of decline for the national status.

### TABLES 5a and 5b

- It was agreed that these tables should be combined into a single table and that this would be better presented as an excel file where filters could be applied to each feature. .
- Information should be presented on a feature by feature basis (as in Table 5a) rather than on a pressure basis (as currently in Table 5b).
- Pressure sources are currently from FEAST but this makes no distinction between which pressures are relevant and which aren't. The most relevant pressures should be highlighted, based on documented evidence in the literature review (Section 3 of the report). This evidence should be put back into the literature review section. Other pressures should be listed, bearing in mind that they may be relevant but have not been studied and so supporting literature may not be available. The source of the pressure information should be presented in the Tables.

### *Application of framework for assessing recovery potential*

#### 1. CASE STUDY DETAILS – UPPER LOCH FYNE and LOCH GOIL

Conservation objective- Conserve (uncertain):

- Burrowed mud
- *Modiolus modiolus*
- *Arctica islandica*

Conservation objective- Recover:

- *Limaria hians*

Activities

- Aquaculture – fin and shellfish
- Fishing – static, diver, demersal (mobile active gear)
- Moorings/anchorage although management is not required as there is no overlap with the species of concern.

Management aims

- Remove activities around sensitive features
- Reduce or limit activity around seapens and fireworks anemone

### Historical information

*Limaria hians* was first recorded in 1988 during otter trawling. At present, the associated community is at the lower end of what would be expected in terms of species richness, abundance and composition. This was previously a mosaic habitat associated with maerl but the maerl has now disappeared. There is no visual evidence of physical damage (although the location of the remaining bed in terms of its depth distribution makes it inaccessible to mobile demersal fishing gear) but the disappearance of maerl (live and dead) is consistent with physical disturbance associated with human activities and this is also likely to have affected the distribution of *Limaria*. There is circumstantial (but questionable) evidence of other pressures, such as freshwater input. Overall, the depth distribution, spatial extent and the integrity of the associated community have been reduced.

In terms of disturbance history, the cause of decline is thought to be a manageable, anthropogenic pressure and the remaining *L. hians* population is patchy in its distribution. Assuming suitable habitat, this implies potential for recolonisation in that a source population exists. However, the density and spatial extent of these patches is not known. There was insufficient information to take this assessment further but pressure removal has been documented to enable recovery of this species and recovery potential is thought to be greatest in areas where some individuals (or patches) remain, rather than uniformly bare habitat. It can be assumed that with effective pressure removal, recovery of *L. hians* in this area will be possible.

*Modiolus modiolus* was present in two locations in Loch Goil in 1989, as a dense bed within the 3-9 m depth range although the status of the bed is not known. In 2010, scattered clumps of *M. modiolus* were recorded with no evidence of the presence of a continuous bed. Density was low. In Loch Fyne, small clumps were found (some near the *Limaria*) in 1988, together with a lot of dead shell. VMS data indicate fishing activity close to this bed. This bed was not re-located in 2010 or 2012 but dead shell was present indicating the possible previous existence of a *Modiolus* bed. There is no information on the associated community. Burrowed mud is present in Loch Fyne and Loch Goil but there is no clear evidence of decline. It is not known whether the distribution of species such as *Pachycerianthus multiplicatus* and *Maxmuellaria lankesteri* would increase in the absence of fishing or whether density and distribution has ever been greater. There is evidence that *M. lankesteri* mounds have been flattened (consistent with damage from mobile demersal fishing gear) but it is not known whether demersal fishing causes direct mortality of this species (or other infaunal species) or whether the mounds support other species which may be compromised in the absence of mounds. However, there is evidence of higher densities of *P. multiplicatus* in unfished areas of Loch Duich and Loch Sween indicating that the Loch Fyne and Loch

Goil communities may be compromised. VMS data indicate that fishing activity occurs in areas of burrowed mud but there is no direct evidence the communities present are any different to what would be expected. This requires further work and the possibility of wider scale data analysis to examine the community structure in fished and un-fished habitats was discussed. That is, are there similarities between fished areas (low density, absence of particular species, small body size) that may be indicative of impacts and is there enough similarity between unfished areas to indicate the characteristics of an unfished area?

### **Application of framework for assessing recovery potential – worked example**

#### ***Limaria hians* – Upper Loch Fyne and Loch Goil**

Assessment of the recovery potential of *Limaria hians* was considered for the Upper Loch Fyne example. Whilst many of the questions relating to the biological aspects of recovery could be addressed, those relating to the cause of the decline could not. Evidence suggests (Minchin, 1995; Trig & Moore, 2009) that this species has the potential to recover following pressure removal. However, in this case, whilst fishing (scallop dredging) is thought to be the cause of the decline, there is no direct evidence. Other information, such as the physical conditions of the habitat and their suitability to support *L. hians* were also unavailable. This hampered the assessment in that if the cause of the decline cannot be addressed, recovery and restoration efforts are likely to be unsuccessful.

<b>Stage 1a Cause of concern</b>		<b>Comments</b>
Is adult mortality occurring?	Yes	Assume mortality based on decrease in spatial extent
Is juvenile mortality occurring?	Yes	Assume mortality based on decrease in spatial extent
Is reproductive activity poor		Not known
Is recruitment occurring		Not known
Has spatial extent of the feature been reduced	Yes	Believe so base on previous records
Have the physical attributes of the habitat changed?		Not known
Is the spatial extent of the physical habitat reduced		Not known
<b>Stage 1b Cause of decline</b>		
Can the decline be linked to anthropogenic pressure/activity (near-field)?	Yes	Assumed to be caused by fishing but no concrete evidence
Can the decline be linked to anthropogenic pressure/activity related (far-field)?		Not known
Can the decline be linked to climate change?		Not known
Can the decline be linked to natural population variability		Not known
Has there been a change in habitat use / spatial distribution?		Not known (unlikely)
To what extent can the decline be linked to disease, parasitism		Not known (unlikely)
To what extent can the decline be linked to competition/interaction with other species		Not known (unlikely)
<b>Stage 1c Disturbance history</b>		
Is the disturbance and the impact patchy with remaining intact patches of habitat/species with some degree of habitat heterogeneity	Yes	An area of <i>Limaria</i> remains (a potential brood stock), possibly because it is inaccessible by mobile benthic fishing gear. No knowledge of the previous status of <i>Limaria</i> in the rest of the area
Is the disturbance widespread and/or associated with total removal of habitat	No	
Has widespread habitat homogenisation occurred		Not known
Concern regarding frequency/intensity/spatial extent of disturbance		Not known – no concrete evidence that fishing is the cause
<b>Stage 1d Potential to remove cause of decline</b>		
Manageable natural pressure		Not known
Unmanageable natural pressure		Not known
Manageable anthropogenic pressure	Yes	Likely to be related to fishing which can be managed
Unmanageable anthropogenic pressure		Not known
<b>Stage 2. Habitat requirements</b>		Various coarse sediments – coarse sand; gravel; mixed muddy gravel; maerl. Depth range predominantly 5-30 m but to a maximum of 100 m. Variable current speeds (weak to strong tidal streams). Semi-infaunal
<b>Stage 3. Factors limiting recovery (Environmental)</b>		

Habitat attributes		Not known
Sediment (physical)		Not known
Sediment (chemical)		Not known
Sediment (biological)		Not known
Hydrodynamic attributes		Not known
Physical/hydrodynamic connectivity		Not known
Water quality		Not known
Appropriate management area		Not known
Residual pressures following management action		Not known
<b>Stage 3. Factors limiting recovery (Biological)</b>		
Initial status of population and status of surrounding, influential populations (brood stock)	Yes	Low density; small brood stock
Biological connectivity		Not known but possibly a problem. No other populations or patches of <i>Limaria</i> were highlighted during the case study presentation
Life history traits	Yes	Limited adult mobility, slow nest development. Damaged individuals susceptible to predation.
Biological interactions		
Poor knowledge of species biology	Yes	
Change in the spatial distribution of the species		Not known
Wider trends in population status		Not known
<b>Stage 4. Possible measures to improve recovery potential</b>		
Pressure removal. Consider near and far-field pressures	Yes	Evidence indicates recovery potential following pressure removal. The recovery process may be prolonged where extensive damage has occurred and/or where populations are fragmented.
Consider change to the management area/scale	Yes	To increase protection of the remaining individuals.
Consider change to the timing of management measures	No	If the decline is related to mobile benthic fishing, a single pass of the gear will cause significant damage. Therefore the timing is not relevant
Consider changes to the activity	Yes	Alternative fishing methods could be considered (diver collection)
Habitat enhancement		Don't know – extent/type of habitat damage is not known
Brood stock protection		
Brood stock protection	Yes	Protection of the existing population plus protection of other, connected populations
Hatchery cultivation		Not known
Translocation		Not known (evidence that bivalves can be translocated but long term success is not known)
Disease control	No	Not thought to be the cause of decline.
Predator control	No	Unlikely to be practical.

### Burrowed mud – Upper Loch Fyne and Loch Goil

Demersal trawling is known to occur in locations where burrowed mud habitats are found but the degree of impact is unclear. There are visual signs of physical modification to the habitat, such as the flattening of *Maxmuelleria lankesteri* mounds but the ecological significance of this is not known (e.g. do these mounds support other species?). Densities of species such a

*Pachycerianthus multiplicatus* are higher in unfished lochs but the relationship between density of this species in Loch Fyne and Loch Goil is not known. For example, other environmental conditions may also influence density and species composition. Because of this, it is difficult to determine the cause for concern, the cause of decline and therefore the potential to remove the cause of decline.

<b>Stage 1a Cause of concern</b>		<b>Comments</b>
Is adult mortality occurring?		No clear evidence of decline. General feeling that the integrity of the feature is compromised
Is juvenile mortality occurring?		No clear evidence of decline. General feeling that the integrity of the feature is compromised
Is reproductive activity poor		Not known
Is recruitment occurring		Not known
Has spatial extent of the feature been reduced		Not known
Have the physical attributes of the habitat changed?		Not known
Is the spatial extent of the physical habitat reduced		Not known
<b>Stage 1b Cause of decline</b>		
Can the decline be linked to anthropogenic pressure/activity (near-field)?		Possibly linked to fishing activity. Evidence that <i>Maxmuelleria</i> mounds are flattened. General feeling that community status would be improved in the absence of fishing but community structure in this area is not well understood. Cannot be sure that it is impacted and cannot categorically link impact to pressure at the present time. Higher densities of typical species are found in unfished areas and fishing has been linked to a decline in the status of this feature in South Arran.
Can the decline be linked to anthropogenic pressure/activity related (far-field)?		Unlikely
Can the decline be linked to climate change?		Not known, unlikely
Can the decline be linked to natural population variability		Not known
Has there been a change in habitat use / spatial distribution?		Not known
To what extent can the decline be linked to disease, parasitism		Not known
To what extent can the decline be linked to competition/interaction with other species		Not known
<b>Stage 1c Disturbance history</b>		
Is the disturbance and the impact patchy with remaining intact patches of habitat/species with some degree of habitat heterogeneity		Not clear but VMS data are available to check
Is the disturbance widespread and/or associated with total removal of habitat		Not clear but VMS data are available to check
Has widespread habitat homogenisation occurred		Not know, not thought to be the case
Concern regarding frequency/intensity/spatial extent of disturbance	Yes	Suspected damage caused by fishing

<b>Stage 1d Potential to remove cause of decline</b>		
Manageable natural pressure	No	
Unmanageable natural pressure	No	Possible but unlikely
Manageable anthropogenic pressure	Yes	Thought to be related to fishing but no concrete evidence
Unmanageable anthropogenic pressure	No	
<b>Stage 2. Habitat requirements</b>		
		Soft muddy substrata. Depth range 10-100m (circalittoral). Very sheltered/sheltered. Weak tidal stream. Full and variable salinity
<b>Stage 3. Factors limiting recovery (Environmental)</b>		
Habitat attributes		Not known
Sediment (physical)		Not known
Sediment (chemical)		Not known
Sediment (biological)		Not known
Hydrodynamic attributes		Not known
Physical/hydrodynamic connectivity		Not known
Water quality		Not known
Appropriate management area		Not known
Residual pressures following management action		Not known
<b>Stage 3. Factors limiting recovery (Biological)</b>		
Initial status of population and status of surrounding, influential populations (brood stock)		Not known
Biological connectivity		Not known
Life history traits		Not well understood but some species are long lived with infrequent recruitment
Biological interactions		Not known
Poor knowledge of species biology	Yes	
Change in the spatial distribution of the species		Not known, not highlighted as an issue
Wider trends in population status		Not known, not highlighted as an issue
<b>Stage 4. Possible measures to improve recovery potential</b>		
Pressure removal. Consider near and far-field pressures		Likely to be effective if the community is degraded
Consider change to the management area/scale		Likely to be effective if the community is degraded – depends what the current scale of management is
Consider change to the timing of management measures		Not known. Avoiding spawning and settlement periods may be beneficial (if they are known) but this will not protect the adults from damage. Therefore, if adult decline is the problem, changes to the timing of management measures won't be effective
Consider changes to the activity		Likely to be effective
Habitat enhancement	No	No evidence that this has been trialled. May be difficult in soft sediment where consolidation, cohesion, sediment sorting and sediment chemistry need to be restored.
Brood stock protection	Yes	Would enhance recovery potential if the community was degraded
Hatchery cultivation	No	No evidence that this has been trialled for the component species of this feature

Translocation	No	No evidence that this has been trialled for the component species of this feature
Disease control	No	No evidence of disease. Disease/parasitism in the component species is unlikely to be well understood and therefore control measures are unlikely to be effective
Predator control	No	Unlikely to be effective

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