Environment Agency

Evidence

Review of existing approaches to evaluate marine habitat vulnerability to commercial fishing activities

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Director of Evidence

Executive summary

The Environment Agency commissioned MarLIN and ABPmer to conduct a review of existing approaches for assessing the sensitivity of marine benthic habitats to fishing impacts. The study evaluated the potential for the methods to be used in developing a Commercial Fisheries Risk Assessment methodology which is required to implement the Water Framework Directive (WFD).

The scope of this desk-based study was to identify and review the physical impacts of fishing activities on marine benthic habitats. Physical impacts arise through direct contact with fishing gears, or from activities associated with fishing, e.g. dropping of weighted pots, anchoring of boats and trampling arising from shore access.

The report reviewed the current state of knowledge on the resistance (intolerance), resilience (recovery) and hence sensitivity of several marine habitats to the effects of commercial fishing activities in the UK. The study identified 130 separate parameters or ecological attributes used in 70 sensitivity assessment methodologies or studies of the effects of fishing activities, together with the links between fishing intensity and habitat response. In addition, the report evaluated the relative strengths and weaknesses of past and current approaches to sensitivity and/or vulnerability assessment worldwide. The following general conclusions were reached:

i. Clear definitions of terms are vital for any assessment procedure. Terms such as resistance, intolerance, resilience recoverability, sensitivity and vulnerability need to be carefully defined and explained.

ii. Habitat groups (derived from the UK marine habitat classification) provide discernable units for assessment that are relatively easy to explain to stakeholders from multiple marine sectors, including the public.

iii. Biogenic habitats and those habitats dominated by long-lived, slow growing species are amongst the most sensitive to damage by fishing activities. Due to their prolonged recovery period, maerl beds may be best viewed as non-renewable resources.

iv. Soft sediment habitats vary in sensitivity depending on the mobility or cohesiveness of the sediment as well as the nature of the communities they support. The rate at which the physical habitat recovers from damage is an important component of the rate at which the habitat as a whole is able to recover.

v. The impacts of fishing activities on chalk reefs habitats are the least well studied of all the habitats examined.

vi. A large number of parameters (130) have been used in the 70 past studies examined to assess the sensitivity of habitats. These range from physical, chemical and biological parameters and include estimators of community structure and function.

vii. No single descriptor or parameter can effectively or reliably explain the impact of fishing on community structure and habitat response. A number of parameters are required to describe the nature of the activity, the nature of the impact or response, the potential rate of recovery and overall sensitivity.

viii. The most used parameters include a suite of biological variables that describe a species or habitat (especially morphology and environmental position), their life history, the physical nature of the habitat itself (especially for soft sediments), and their contribution to ecosystem function (e.g. biogenic habitats, biomass and productivity).

ix. However, biological traits alone cannot necessarily capture all aspects of the sensitivity of marine habitats, due to lack of data and understanding, and there is an important role for expert judgement in the assessment procedure.

x. Recent meta-studies and empirical studies of the effects of fishing on marine habitats (primarily soft sediments) have improved our understanding of the relationship between fishing intensity, gear type, substratum type and impact and recovery.

xi. Studies of the effects of different fishing intensities are underpinned by Vessel Monitoring System (VMS) data on the movement of fishing vessels.

xii. The setting of clear, well defined, thresholds is a vital part of the assessment procedure. Thresholds include definitions of fishing intensities and gear types and thresholds of damage (acceptable vs. unacceptable), scales of resistance, resilience, sensitivity and vulnerability.

xiii. The sensitivity assessment methodologies reviewed were all developed for specific purposes, i.e. to answer specific management questions. Thus, they are not completely applicable outside their original design parameters.

xiv. Sensitivity assessment is designed to manage uncertainties and information gaps. Although our understanding of the effects of fishing has grown considerably over the last twenty years, information gaps remain.

xv. The existing sensitivity assessment methodologies provide a wide range of tools that could be applied to vulnerability assessment. Therefore, the development of an approach to the assessment of the vulnerability of habitats to commercial fishing activities is feasible.

The following recommendations follow from these conclusions.

i. The development of risk assessments for commercial fisheries activities within WFD requires a targeted approach that builds on existing expertise.

ii. Any sensitivity assessment procedure needs a clear definition of the management questions it is designed to answer, the decisions it is designed to support, the scale at which it is to be applied and hence its limitations outside that remit.

iii. Engagement with other agencies and their approaches is a potentially cost-effective way of drawing on a range of expertise, producing a widely supported methodology while reducing replication of effort.

iv. The terms ‘resistance’ and ‘resilience’ should be used in preference to ‘intolerance’ or ‘recoverability’; and ‘risk’ in preference to ‘vulnerability’.

v. The approach developed should include a systematic approach to assess ‘sensitivity’ and ‘risk’ followed by expert validation.

Overall, there is enough data, evidence and expertise to develop a systematic approach to the assessment of the vulnerability (and risk) of marine habitats to commercial fishing activities. But, before a methodology can be developed, clear decisions need to be made about the management and conservation questions the approach needs to answer, and the user group (or stakeholder group) that will need to understand and implement management.

Acknowledgements

The project team would like to thanks Dr Cristina Vina-Herbon for visualising and initiating this study.

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1 Introduction

A wide range of studies have been taken forward over the last 10 years to improve scientific understanding of the relative sensitivity of marine organisms and habitats to various human pressures and natural events, for example, MacDonald et al. (1996); Hiscock et al. (1999); Laffoley et al. (2000); Hiscock and Tyler-Walters (2006). The concepts of sensitivity and vulnerability have been discussed and carefully defined (Hiscock 1999; Hiscock et al. 1999; Laffoley et al. 2000) as have comparable definitions such as resistance and resilience (Hall et al. 2007; Robinson et al. 2008). The assessment of the relative sensitivity, and hence vulnerability, of marine species, habitats or landscapes has long been held as a potentially powerful tool in marine environmental management and planning at local, regional and national scales.

In general, assessments have considered that sensitivity is a measure of the degree to which a receptor is affected by an impact and the ability of the receptor to recover from this (although studies may only focus on one aspect). Other studies have used the terms 'resistance' and 'resilience' to encompass similar concepts. The variety of studies and the differing purposes of these, and hence their approaches, have meant that alternative definitions and measures of sensitivity have been used in marine habitat assessment and other fields.

The Environment Agency commissioned MarLIN and ABPmer to conduct a review of existing approaches to assessing sensitivity of marine benthic habitats to fishing impacts to support the Commercial Fisheries Risk Assessment (part of the implementation of the Water Framework Directive, WFD).

In this review, both aspects of sensitivity have been considered. The first is defined as 'resistance' and the second as 'resilience'. Alternative definitions of terms are discussed in the review. Many of the terms and assessments considered in the review can be applied to habitats or species; we have therefore used the term receptor throughout and made explicit where specifically, habitats or species are being considered.

The specific tasks that this review was intended to address were:

Task 1: Describe and evaluate the current understanding of habitat response to commercial fishing activities in terms of resistance and resilience (Section 3, Appendix 1);

Task 2: Develop a list of habitat parameters and/or ecological attributes that have been used in previous publications to evaluate or measure morphological pressures and an evaluation of their effectiveness in assessing impacts from fishing activities (Section 4, Appendix 2);

Task 3: Evaluate the current information linking fishing frequency and habitat response and an assessment of the use of thresholds to identify habitat sensitivity (Section 5);

Task 4: Compare different approaches to assessing sensitivity and vulnerability (Section 6)

Task 5: Identify knowledge gaps in terms of habitats and species, fishing activities and geographic locations (Section 7).

Task 6: Collate the information gathered through Tasks 1-5 into an Excel spreadsheet. This is supplied separately but sample outputs are can be found in Appendix 1 and 2

Task 7: Develop a feasibility assessment and a list of recommendations based on the review to support further work in quantifying vulnerability.

2 Methodology

The review was primarily a desk study built on the prior experience of the authors. For example, ABPmer have undertaken a number of previous studies exploring the potential sensitivities of protected habitats and species to renewable energy development (ABPmer 2005, 2006, 2010 (in prep)). Similarly, the Marine Life Information Network (MarLIN) developed an approach to sensitivity assessment for Defra in 1999, and have been responsible for the production of reviews of sensitivity of most of the UK's priority marine habitats and species, and applied sensitivity assessment at a variety of scales (from species and biotopes to landscapes). Therefore, MarLIN has particular experience and knowledge of sensitivity assessment.

2.1 Defining terms

The literature on 'sensitivity' assessment uses a number of seemingly synonymous terms. However, many studies use the term 'sensitivity' in different ways, and sensitivity *sensu stricto* (i.e. as defined) does not always equate to how the term is perceived. Therefore, it is important to define the terms as used in this report, with respect to prior use (see Table 1 overleaf).

2.1.1 Defining 'Sensitivity', 'Resistance' and 'Resilience'

Sensitivity has been defined as 'the innate capacity of an organism to suffer damage or death from an external factor beyond the range of environmental parameters normally experienced' (Holt et al. 1995). This definition is widely accepted (McLeod 1996, Tyler-Walters et al. 2001, Zacharias and Gregr 2005), and has been extended beyond the focus on single organisms to include 'the …*habitat, community or species*' (McLeod 1996). Sensitivity therefore encompasses a measure of the effect of a pressure (sometimes referred to as disturbance, perturbation, impact, effect or stress), on a receptor (see Table 1 for definitions of key terms).

Sensitivity is typically defined or measured in terms of two aspects, the ability of a receptor to withstand an impact and to recover from it (Table 1). These attributes were described by Holling (1973) for systems in general, where the term 'resistance' refers to the ability to absorb disturbance or stress without changing character and the term 'resilience' describes the speed at which the system returns to its previous state when changed. Other studies have referred to 'resistance' as 'tolerance' or 'intolerance'. 'Resilience' can be thought of as synonymous with the ability of a system to recover from a perturbation, which some studies have referred to as 'recoverability' (Holt et al. 1997).

In the UK Review of Marine Nature Conservation (RMNC), Laffoley et al. (2000), defined sensitivity as '*dependent on the intolerance of a species or habitat to damage from an external factor and the time taken for its subsequent recovery'*. The Oslo and Paris commission (OSPAR) also uses both of these concepts to evaluate sensitivity as part of the criteria used to identify 'threatened and declining' species and habitats within the OSPAR region - the Texel-Faial criteria. A species is defined as '*very sensitive*' when it is '*easily adversely affected by human activity (low resistance) and/or it has low resilience*' (recovery is only achieved after a prolonged period, if at all). Highly sensitive species or habitats are those with both low resistance and resilience.

Table 1 Definition of sensitivity and associated terms

Term	Definition	Sources
Sensitivity	A measure of tolerance (or intolerance) to changes in environmental conditions	Holt et al. (1995), McLeod (1996), Tyler-Walters et al. (2001), Zacharias and Gregr 2005)
Resistance (Intolerance)	Resistance characteristics indicate whether a receptor can absorb disturbance or stress without changing character.	Holling (1973)
Resilience (Recoverability)	The ability of a receptor to recover from disturbance or stress.	Holling (1973)
Vulnerability	Vulnerability is a measure of the degree of exposure of a receptor to a pressure to which it is sensitive.	Based on Hiscock et al. 1999; Oakwood Environmental Ltd (2002).
Pressure	The mechanism through which an activity has an effect on any part of the ecosystem. The nature of the pressure is determined by activity type, intensity and distribution.	Robinson et al. (2008)
Impact	The effects (or consequences) of a pressure on a component.	Robinson et al. (2008)
Exposure	The action of a pressure on a receptor, with regard to the extent, magnitude and duration of the pressure.	Robinson et al. (2008)

2.1.2 Defining 'Vulnerability'

A habitat, community or species becomes 'vulnerable' to adverse effect(s) when it is sensitive to an external factor (pressure or activity) and that external factor is likely to affect the habitat, community or species (Holt et al. 1995, Tyler-Walters et al. 2001, Oakwood Environmental Ltd 2002). Vulnerability can therefore be considered to be a measure that combines information on sensitivity and exposure to an impact (Table 1).

If a receptor is not sensitive to a pressure then it is not vulnerable, equally, if it is not exposed to an impact it is not vulnerable. So that, while, a certain habitat type may be highly sensitive to fishing activities, if it occurs in an area where there is never any fishing activity it would not be vulnerable. Alternatively, a habitat that is less sensitive to fishing activities, that is in an area where it is repeatedly exposed to fishing, is vulnerable to some degree.

As the intensity and/or duration of the impact (the exposure) usually determines the magnitude of effect, measures of vulnerability often take into account the probability of an impact and the probable characteristics of impacts, i.e. by classing vulnerability according to different intensity regimes (Oakwood Environmental Ltd, 2002). The vulnerability rating indicates the likely severity of damage should the pressure occur at a defined intensity and/or frequency.

2.2 Literature review and information sources

Although the project team had almost a decade of experience in the area at their finger-tips, a short additional review of material was undertaken to fill gaps and ensure the most recent developments were captured.

The library services of the National Marine Biological Library (NMBL, Plymouth) and ABPmer Ltd were used. The review of primary scientific literature, and grey literature utilised the following resources:

- the literature database Scopus;
- the MarLIN Biology and Sensitivity Key Information database;
- the NMBL library catalogue
- Aquatic Sciences and Fisheries Abstracts (ASFA);
- Web of Science (ISI); and
- Google Scholar.

A number of search terms were used including ‘sensitivity marine’, ‘recovery marine’, ‘risk assessment’, ‘environmental sensitivity indices’ and ‘sensitivity index’.

The review paid particular attention to sensitivity assessment approaches designed for fishing impacts. However, in order to evaluate the full range of approaches that could be applied to the impacts of commercial fishing activities, it was necessary to examine a wide range of approaches from the UK, Europe and internationally. In addition, studies that used biological traits analysis to detect the effects of fishing impacts were also examined to inform the section on ecological attributes (parameters).

2.2.1 Review of the Impacts of Commercial Fishing Activities on Marine Habitats (Task 1)

The scope of this study was to identify and review the physical impacts of fishing activities on marine benthic habitats. Physical impacts arise through direct contact with fishing gears, or from activities associated with fishing, e.g. dropping of weighted pots, anchoring of boats and trampling arising from shore access.

The impacts of commercial fishing activities on the habitat groups were assessed in terms of resistance (tolerance) and resilience (recovery) and this information is presented for each of 12 habitat groups based on habitat types supplied by the Environment Agency (see Section 2.2.2 and Table 2 overleaf).

For each group physical impacts on three habitat components were considered:

i. Biological features which were defined as individuals and the populations that these are part of;

ii. Biogenic features- the habitat structures that are created by organisms, including structures such as reefs or attachment structures that are formed from the bodies of individuals (so that this is closely related to the first component);

iii. Geomorphological / sedimentary features of habitats including the substrate and associated forms e.g. sand ripples.

The degree of fishing impacts on each of these will vary between habitat groups and these therefore provide useful categories to discriminate the sensitivity of different habitat types. It should be noted however that there is some overlap between categories. Resistance and resilience of biological features, for example, will be closely linked to resistance and resilience of biogenic features.

The reviews were intended to be concise and therefore only effects on numerically dominant or characteristic species were considered. References for review papers and sensitivity assessments are also provided. Indirect effects on habitats such as changes in ecological functioning and food webs that occur as a result of changes in species composition arising from fishing activities were considered outside the scope of this review.

The reviews are informed by empirical observations on the physical impacts of fishing activities on marine habitats. In general, life history and ecological traits of individuals determine sensitivity to fishing impacts. Attached epifauna and flora and shallow dwelling, sedentary infauna are unable to move away from gears and cannot avoid being hit if they are in the direct path of fishing gear, weights, anchors, rope, feet or vehicles. The degree of damage will depend on the fragility or robustness of the organism. Flexible or thick-shelled organisms may survive a hit that will damage or kill a brittle, inflexible organism, such as an echinoderm or a bivalve.

Larger species are more vulnerable than smaller species to towed gears (Bergman and van Santbrink 2000a, Bergman and van Santbrink 2000b). Nets and ropes may pass over smaller organisms or pressure waves in front of towed gears may push smaller organisms out of the way. Body size and life history are linked; larger organisms have lower natural mortalities, slower growth and lower annual reproductive output, increased longevity and lower annual reproductive output, increased longevity and lower natural rates of intrinsic increase (Brey 1999). This means that such species are predicted to be less able to compensate for high mortality rates inflicted by fishing.

General predictions can also be made about the severity of fishing impacts based on habitat type. Prevailing physical and chemical conditions play a large role in determining the biological assemblage that is present at a location. In areas that are subject to high levels of natural disturbance such as high wave action and dynamic sediments, the assemblage present will be composed of small, robust organisms able to withstand or recover quickly from, disturbances. Conversely, assemblages that develop in stable habitats are likely to contain organisms that are intolerant of disturbances and recover only slowly. In sheltered locations, the absence of short-term perturbations may allow structurally complex habitats, containing larger and longer-lived animals, to develop. A number of reviews have shown that these complex habitats are more impacted by fishing activities and slower to recover (Auster 1998, Kaiser et al. 2006).

2.2.2 Habitat Groups

Habitats and biotopes are ecological units that are frequently used in sensitivity assessments. Marine habitats are typically classified into types using attributes such as sediment type or the dominant species present or through hierarchical classification schemes that combine information on the habitat and biological assemblage such as the UK Biotope classification (Connor et al. 1997a, Connor et al. 1997b, Connor et al. 2004). Efforts are being made to ensure consistency in classification, particularly between UK and European (EUNIS) classification systems.

It should be recognised that, given natural variability, classification schemes represent an artificial sub-division and that some difficulties may arise through the lack of a single standardised classification system. For example, a journal paper may simply refer to an experiment to determine fishing impacts on a ‘sandy’ habitat. Within the UK biotope classification scheme, this could refer to one of 17 biotopes occurring on sandy sediments, where the biological assemblage and environmental conditions are relatively distinct between biotope types.

For the purposes of this review, the Environment Agency provided a list of 29 marine habitat types (EUNIS level 4). These have been grouped for previous studies by the Countryside Council for Wales (CCW) (Hall et al. 2008) and for this task were further grouped into 12 broad habitat categories based primarily on physical similarities (sediment) and the biological assemblage (Table 2).

Table 2 Habitat categories used in this review and the corresponding CCW habitat groups*

Habitat Groups	CCW Habitat Groups*
Biogenic reefs	H6, H17
Bivalve reefs	H4, H5, H25, H27,
Sand sediments	H18, H19, H24
Mud sediments	H10,H11, H12, H13, H14
Mixed sediments	H28, H29
Eelgrass	H30
Saltmarsh	H13
Macroalgae dominated subtidal hard substrates	H22, H23
Faunal turfs (bryozoans/hydroids)	H20
Slow growing epifauna	H15, H21
Vertical and underboulder surfaces	
Chalk reefs	
Mixed sediment	H28,H16, H29

* Based on information supplied by the Environment Agency

2.2.3 Review of Habitat Parameters / Ecological Attributes used in Previous Approaches (Task 2)

The numerous approaches to the assessment of habitat ‘sensitivity’ have used a number of habitat and species parameters. For example, physical parameters are used in the approach of Gundlach and Hayes (1978) as applied to oil spill sensitivity, while biological characteristics are not included. Over the last ten years, there has been increased interest in including biological characteristics, either of the species that dominate (or characterize) the habitat, or of the community interactions, within the associated community.

In this review, prior approaches to sensitivity assessment were examined to identify the parameters or attributes of the component species or habitats that were used. In this review, the term parameter is taken to mean *a quantified or qualified characteristic of a species, a habitat or its associated community.* For example, a species may be quantified by its abundance, its size or salinity preferences but qualified by its shape, growth form or habit. A habitat may be quantified by its substratum (sedimentary grain size, or bedrock), extent (area) or depth and qualified by its dominant flora or fauna. The community may be quantified by productivity and qualified by the nature of its community interactions.

In addition to prior approaches to sensitivity assessment, the review also looked at studies of the effects of fishing activities on marine habitats, in order to identify those parameters that are most effective at describing effects and hence sensitivity.

2.2.4 Review of the current information linking fishing frequency and habitat response (Task 3)

Recent advances in VMS (Vessel Monitoring System) technology and their application to fishing vessels has provided scientists with the opportunity to map fishing intensity and examine relative habitat sensitivity (Collie et al. 2000, Hiddink et al. 2006, Mills et al. 2006, Eastwood et al. 2007, Dunstone 2009). A short review of recent information was undertaken.

In addition, the review looked at information used by prior approaches to set thresholds of impact or effect, and their practicality for sensitivity assessment.

3 Impacts of Commercial Fishing Activities on Marine Habitats

3.1 Biogenic Reefs

The habitat types (and corresponding CCW habitat groups and EUNIS classification) referred to in this review section are shown in Table 3.

Table 3 Biogenic reef habitat groups

Habitat types	CCW habitat group	EUNIS Level 3	EUNIS Level 4	EUNIS Name
Sabellaria worm reefs	H6	A2.7	A2.71	Littoral *Sabellaria* reefs
Maerl beds	H17	A5.5	A5.51	Maerl beds
	H17	A3.1	A3.14	Encrusting algal communities

3.1.1 Brief Description of Habitat Group(s):

Biogenic reefs can be defined as solid, massive structures which are created by accumulations of organisms, usually rising from the seabed, or at least clearly forming a substantial, discrete community or habitat which is very different from the surrounding seabed (Brown et al. 1997). The structure of the reef may be composed almost entirely of the reef building organism and its tubes or shells, or it may to some degree be composed of sediments, stones and shells bound together by the organisms (Holt et al. 1998)

Biogenic reefs are generally subtidal but may extend to the intertidal zone. Only a few species are able to develop biogenic reefs; for the purposes of this review, biogenic reefs created by the following species have been considered: *Sabellaria alveolata*, *S. spinulosa* and maerl species and these are discussed briefly below. Biogenic reefs formed by the species *Modiolus modiolus* and *Mytilus edulis* have been considered in the review of the Bivalve Beds (Section 3.2).

3.1.2 Honeycomb worm *Sabellaria alveolata*

The sedentary polychaete *Sabellaria alveolata* (honeycomb worm) builds tubes from sand and shell. On exposed shores, where there is a plentiful supply of sediment, *S. alveolata* can form dense aggregations, which may be regarded as reefs on boulders and low-lying bedrock (Connor et al. 2004).

The range of *S. alveolata* reefs is essentially southern and western Britain and the reefs are usually intertidal, although subtidal reefs have been reported. They are generally limited to areas of hard substratum, including cobble, adjacent to sand and with moderate to considerable exposure to waves (Holt et al. 1998).

Intertidal *S. alveolata* reefs are not particularly diverse communities, though they do nevertheless provide some increased diversity of habitat. Sheets of *S. alveolata* appear to enhance algal diversity, apparently by providing barriers to limpet grazing (Cunningham et al., 1984). Older reefs have somewhat more diverse associated communities than younger ones as they provide a variety of habitats for other species, often in crevices (Holt et al. 1998).

Sublittoral *S. alveolata* reefs occur on tide-swept sandy mixed sediments with cobbles and pebbles and are considerably less extensive than the intertidal reefs formed by this species. The presence of *Sabellaria* sp. has a strong influence on the associated infauna as the tubes bind the surface sediments together and provide increased stability. Other associated species may include polychaetes and amphipods.

3.1.3 Ross worm *Sabellaria spinulosa*

Another sedentary polychaete *Sabellaria spinulosa*, which is commonly reported to be found in solitary form, may qualify as biogenic reef community on the basis of their strong alteration of habitat, although this species rarely forms substantial raised areas. The worms and tubes themselves are rather smaller than those of *S. alveolata*, and structured, 'honeycomb' like arrangements have never been reported for *S. spinulosa* (Holt et al. 1998).

In the subtidal, *S. spinulosa* is found on mixed sediments in a variety of hydrographic conditions and typically forms loose agglomerations of tubes forming a low lying matrix of sand, gravel, mud and tubes on the seabed. The infauna comprises of typical sublittoral polychaete worm species and cirratulids, together with bivalve molluscs, and tube building amphipods. The epifauna comprise a variety of bryozoans (sea mats and horn wracks) in addition to calcareous tubeworms, pycnogonids (sea spiders), hermit crabs and amphipods. The reefs formed by *S. spinulosa* consolidate the sediment and allow the settlement of other species not found in adjacent habitats leading to a diverse community of epifaunal and infauna species (Connor et al. 2004).

3.1.4 Maerl

Maerl develops when coralline red algae, which have a hard calcium carbonate skeleton, become free-living (not attached to rock or pebble substratum) due to fragmentation. The 'loose-lying' maerl sometimes accumulate into flat beds, ripples or large banks of live and dead maerl, or dead maerl only, with or without terrigenous material (Birkett et al. 1998a). The three-dimensional structure of maerl forms an interlocking lattice that provides a wide range of niches for infaunal and epifaunal invertebrates. Hence maerl habitats support a high diversity and abundance of species, including a small group of species which are confined to maerl habitats (rarely found elsewhere) and other invertebrates and algae that are found predominantly on maerl (Birkett et al. 1998a).

Maerl beds are confined to a small proportion of European shallow sublittoral waters and are patchily distributed around the UK coast. Maerl beds are found in areas characterised primarily by high water movements in the photic zone (Kamenos et al., 2003) and occur on a very broad range of underlying substrata (Birkett et al. 1998a). Within the UK, Scotland is home to the most extensive maerl beds in Europe (Birkett et al. 1998a).

Three main species of free-living coralline algae are reported to occur in European waters, with a further six to eight species known to contribute to deposits in certain areas. In southern Britain, maerl beds consist of *Phymatolithon calcareum* and

Lithothamnion corallioides. *Lithothamnion corallioides* is replaced in Scotland by *L. glaciale* (Hall-Spencer 1995). *Phymatolithon calcareum* is both the most widely distributed and the most abundant maerl species in the UK.

Maerl species are very slow-growing algae and maerl biotopes are fragile according to most recognised categories of fragility and are currently threatened by several anthropogenic activities including the use of heavy demersal fishing gear (Birkett et al. 1998a).

3.1.5 Description of Fishing Activities

The commercial fishing activities likely to occur within these habitat groups are shown in Table 4.

Table 4 Commercial fishing activities associated with biogenic reefs

Habitat Group	Shore Access	Hand Gathering	Static Gears	Towed gears
H6	✓		✓	✓
H17			✓	✓

Trampling and shore access to fishing grounds may damage intertidal *Sabellaria alveolata* reefs (Tyler-Walters and Arnold 2008), whilst bait digging (removal of the *S. alveolata* worms) has been identified as another possible impact on intertidal reefs although this has not been observed at an intensive scale (Holt et al. 1998). Subtidal *S. alveolata* and *S. spinulosa* reefs may be affected by the use of static and towed fishing gears.

Northern European maerl beds typically occur in shallow (<32 m) waters where there are high rates of water exchange. This encourages the growth of an abundance of epifaunal and infaunal bivalves including scallops (*Aequipecten* spp., *Pecten* spp.), razor clams (*Ensis* spp.) and clams (*Dosinia* spp., *Tapes* spp.) making maerl habitats attractive to fishers (Hauton et al. 2003). In addition, gadoids may be attracted by the substratum heterogeneity (Kamenos et al. 2003). Towed demersal fishing gears that may be used in this habitat group include shrimp trawling, scallop dredging and hydraulic dredging (for bivalves).

3.1.6 Reviews and Sensitivity Assessments

Numerous studies have assessed the impacts of fishing activity (and in-particular towed demersal gear) on the biogenic reefs being considered in this section, including: impacts on maerl beds (Hall-Spencer and Moore 2000b, a, Bordehore et al. 2003, Hauton et al. 2003, Kamenos et al. 2003) and impacts on *Sabellaria* reefs (Holt et al. 1997, Vorberg 2000).

Tyler-Walters and Arnold (2008) assessed the sensitivity of *Sabellaria alveolata* reefs to impacts caused by access to fishing grounds. Macdonald et al (1996) reviewed the sensitivity of seabed types and benthic species, including maerl and *Sabellaria* spp., to a ‘single encounter’ with static and towed fishing gears. More recently, the sensitivity of subtidal biogenic reefs to fishing impacts was assessed by Hall et al. (2008).

The Marine Life Information Network (MarLIN) has produced sensitivity assessments using the MarLIN approach (including factors relevant to fishing activities) for *Sabellaria alveolata, Sabellaria spinulosa* and maerl. They have also assessed the sensitivity of a number of habitat types in which biogenic reefs occur including:

- *Sabellaria spinulosa*, didemnids and other small ascidians on tide-swept moderately wave-exposed circalittoral rock;
- *Sabellaria alveolata* reefs on sand-abraded eulittoral rock Sabellaria alveolata reefs on sand-abraded eulittoral rock; and
- *Sabellaria spinulosa* with kelp and red seaweeds on sand-influenced infralittoral rock.

MarLIN assessments of sensitivity (intolerance and recoverability) are summarised in Appendix 1.

3.1.7 Resistance (Tolerance)

Maerl

Towed gears have been shown to damage, remove and bury maerl. In areas of high fishing intensity maerl beds may be replaced by faster growing, more resistant, non-coralline algae altering the habitat type present and conservation value. Fishing reduces the species richness, abundance and biomass of associated macrofauna. Studies have shown that the impact of commercial fishing activities depends on the intensity of the fishing activity and the gear type used.

The impact of using a hydraulic blade edge on maerl beds was investigated by Haunton et al. (2003) in the Clyde Sea on the west coast of Scotland. The experimental hydraulic dredging removed, dispersed and buried the maerl (fluorescently labelled dead maerl was used as a proxy for live maerl) at a rate of 5.2 kg/m^2. A small proportion of the maerl was retained in the dredge; the remaining maerl was either smashed and dispersed along the dredge track or ploughed into the seabed.

Preliminary work on Maltese maerl beds indicate that commercial otter trawling has had no negative impact on the cover of live maerl thalli (BIOMAERL team, 1999, cited in Hall-Spencer and Moore 2000b)

Hall-Spencer and Moore (2000a, 2000b) investigated the immediate and long-term impact of scallop dredging in maerl beds. The study showed that a single tow of three Newhaven scallop dredges resulted in live maerl being buried up to 8 cm below the sediment surface and biogenic carbonate structures (maerl thalli, echinoid test plates and bivalves shells) being crushed and compacted.

Bordehore et al. (2003) compared the associated fauna of maerl beds in areas with low and high frequency of trawling activity. In the low frequency area, the cover of rhodoliths was four times greater and the mean maximum size of rhodolith was larger compared to the high frequency trawl area. There were also significant differences in the algae community of the two areas, with 50 per cent of the algal cover comprising Corallinales in the low frequency trawled area compared to 90 per cent cover of non-Corallinales algae in the high frequency trawled area. Species richness, density and the biomass of macrofauna was higher in the low frequency trawl area compared to the high frequency area.

In general, benthic communities are relatively unaffected by static fishing gears (pots, long-lines or anchored nets) due to the relatively small area of seabed directly affected (Kinnear et al. 1996, Jennings and Kaiser 1998, Eno et al. 2001). Benthic community biomass in areas subjected to only static gear use has been reported to be significantly greater compared to areas in which trawling has occurred within the last two years (Blyth et al. 2004).

In an assessment of the sensitivity of seabed types and benthic species to a 'single encounter' with static and towed fishing gears, MacDonald et al. (1996) calculated that maerl was highly sensitive to encounters with high impact fishing gears (e.g. scallop dredge) due to being very fragility and having a long recovery time (see also Resilience, Section 3.1.10).

Sabellaria Species

Shore access to fishing grounds may lead to trampling damage to *Sabellaria alveolata* reefs. Damage depends on the intensity and behaviour of the pedestrians, and the impacts vary from minor damage to tubes, to the production of cracks and removal of reef sections (Cunningham et al. 1984).

Sabellaria spinulosa may be fairly resistant to light trawling. A study by Vorberg (2000) suggested that shrimp trawlers (regarded as relatively light fishing gear) trawled over a *Sabellaria spinulosa* reef stirred up the top sediment layer, producing clouds of fine grain material but was not observed to damage the reef structure itself, although it was noted that the study only addressed short-term effects of a once-only disturbance. The highest load-bearing capacity (mean compressive strength value before fracture) for the upper part of the reef block (which would be exposed to trawling) was 2.2 kg/cm^2.

In an assessment of the sensitivity of seabed types and benthic species to a 'single encounter' with static and towed fishing gears, MacDonald et al (1996) calculated that *S. alveolata* and *S. spinulosa* reefs were moderately sensitive (scoring 50 out of a maximum sensitivity score of 100; with 100 representing the highest sensitivity) to encounters with high impact fishing gears. The overall sensitivity of *Sabellaria* reefs was lower compared to maerl, despite also being categorised as very fragile, due to its shorter recovery time (see Resilience, Section 3.1.10).

3.1.8 Biogenic Features

Structurally complex habitats, including biogenic reefs, are more adversely affected by fishing activities compared to unconsolidated sediment habitats that occur in shallow waters (Kaiser et al. 2002). This is due to the damage/removal of organisms (e.g. high biomass emergent species and/or species producing biogenic structures) that provide a three-dimensional habitat for other animals.

3.1.9 Geomorphological /Sedimentary Features

Maerl Beds

Towed fishing gears can alter the topography and structure of sediments, even at low fishing intensities. For example, a single pass of a hydraulic dredge for bivalves in maerl beds has been reported to alter sediment structure up to 9cm below the seabed surface, significantly reducing the gravel component. Hence, repeated dredging at fishery scale would be expected to alter the physical structure of the sediment producing a sandier habitat (Hauton et al. 2003). The same study showed that another impact of dredging was the smothering of the surrounding maerl with suspended sediment (a 20-fold increase in the amount of sediment settling around the dredge was recorded).

Scallop dredging has also been shown to alter the sediment composition of maerl grounds (Bordehore et al., 2003), reducing the substratum heterogeneity and creating an area resembling a gravel bottom in structure (Kamendos et al., 2003). Reductions in the heterogeneity of maerl beds would be expected to reduce biodiversity, juvenile settlement (high substratum heterogeneity is attractive as nursery areas), and later, recruitment (Kamendos et al., 2003).

The physical impact of scallop dredging in maerl grounds was described in Hall-Spencer and Moore (2000b, a). A single tow of three Newhaven scallop dredgers resulted in significant physical disturbance along a 2.54 m wide transect. The sculpted ridges and troughs of the dredge tracks remained visible for 2.5 years at one previously unfished study site and for 1.5 years at one site that was commercially fished. The rate of sediment erosion was calculated as 340 g/m of dredge track and the redistributed sediment blanketed an area up to at least 15 m away from the dredge path. There was also a change in the granulometric structure of the surface sediment compared to adjacent undredged areas, with loss of vertical stratification and open lattice layers, less interstitial space and higher proportion of fine particles at the surface.

3.1.10 Resilience (Recovery)

Resilience appears to vary between different biogenic reef habitat types. For example, Hall-Spencer and Moore (2000b, a) found that five months after a single tow of a scallop dredger there were 70-80 per cent fewer live maerl thalli compared to prior to the test dredge. The dredging had caused maerl thalli to be buried within the sediment and therefore killed through lack of light. There were no discernable signs of recovery, either in numbers or area covered by live maerl thalli, over the four year recovery monitoring period, due to the slow growth and poor recruitment of maerl species.

Intertidal and subtidal *Sabellaria* reefs however may recover more rapidly from minor damage due to rapid growth rates (12 cm/year cited in McMath et al. 2000). More severe damage, such as removal of reefs will require longer recovery times and may depend on recolonization events. MacDonald et al (1996) estimated that *Sabellaria* reefs were less sensitive to all types of fishing gears (i.e. low, medium and high impact gears) than maerl, due to the shorter recovery time of *Sabellaria* spp. (based on assessments of a ‘single’ fishing disturbance followed by a recovery period with no fishing activity).

Biogenic Features

Biogenic habitats, in general, have the longest recovery trajectory with respect to recolonization of the habitat by its associated fauna (Kaiser et al., 2002). Kaiser et al. (2006) undertook a meta-analysis to examine the response and recovery of benthic biota in different habitats to different fishing gears. They found that scallop dredging in biogenic habitats had the most severe initial impacts compared to deployment in other habitats, with no evidence of recovery post scallop dredging within the recovery time periods of the studies analysed (the longest post-impact period included in data set was 1,460 days).

The settlement of *Sabellaria* larvae is encouraged by the presence of con-specifics, so that reef removal in an area due to trawling may inhibit the establishment of a replacement colony.

Geomorphological/Sedimentary Features

The biogenic reefs discussed occur on mobile rather than hard substrates. The changes in the granulometric structure of the surface sediment in maerl grounds caused by a single scallop dredge tow, gradually returned to the baseline state of a clean gravelly upper layer of maerl over a four year monitoring period (Hall-Spencer and Moore 2000b). The authors concluded that this was presumably due to the winnowing away of the fine materials, brought to the surface by the action of the scallop dredge, by water movement. Local hydrographic and sediment conditions therefore, influence recovery rates for this habitat type.

3.2 Bivalve Beds

The habitat types (and corresponding CCW habitat groups and EUNIS classification) referred to in this review section are shown in Table 5.

Table 5 Bivalve beds habitat groups

Habitat types	CCW habitat group	EUNIS L3	EUNIS L4	EUNIS Name
Rocky, gravely littoral, Mussel beds	H4	A1.1	A1.11	Mussel and/or barnacle communities
	H4	A1.1	A1.22	Mussels and fucoids on moderately exposed shores
	H4	A2.2	A2.21	Strandline
Mussel beds (exploited) Subtidal	H4, Subtidal	A4.2	A4.24	Mussel beds on circalittoral rock
	H4, Subtidal	A5.6	A5.62	Sublittoral mussel beds on sediment
	H4, Subtidal	A3.1, A 3.3		
Rocky, gravely littoral, Mussel beds	H5	A1.1	A1.49	Hydro-littoral mussel beds
	H5	A2.7	A2.72	Littoral mussel beds on sediment
Oyster beds (exploited)	H25	A2.1	A2.12	Estuarine coarse sediment shores
	H25	A5.6	A5.64	Pontic *Ostrea edulis* biogenic reefs on mobile sea bottom
	H25	A5.4	A5.43	Infralittoral mixed sediments

3.2.1 Brief Description of Habitat Group(s):

These habitat groups are characterised by relatively large and long-lived bivalves. In many situations, habitat graduations can be seen from those where the bivalves are scattered individuals, to a patchy extent without building substantial mounds, to those where the beds form substantial reefs. Species include the edible oyster *Ostrea edulis*, common mussels *Mytilus* edulis and horse mussels *Modiolus modiolus* and these species are considered in this section. Bivalve beds can occur on either soft or hard substrates, intertidally or subtidally. In some cases, the beds can form reef structures although the form of these varies between species. The horse mussel *Modiolus modiolus* is mainly infaunal with typically, only a short part of the shell ends protruding from the surface. The byssus threads stabilise sediments and large amounts of dead shell can be present.

Mytilus edulis can be abundant all over the UK in intertidal and sometimes subtidal habitats, ranging from fully saline to highly estuarine; over much of this range it is capable of forming dense beds (Holt et al. 1998). *Mytilus edulis* reefs are comprised of living and dead mussel shells at high densities, bound together by byssus threads, and incorporate a large amount of accumulated sediment, faeces and pseusdofaeces. The beds are therefore high in organic matter. Well developed reefs in most UK sites rarely exceed 30-50 cm (Holt et al. 1998). Mussel bed thickness and structural complexity increase with age of the bed.

The bivalve beds can stabilise sediments and provide food, attachment space and refugia for a number of species. Such reefs are therefore associated with high species richness.

Bed longevity varies, some reefs may be short lived while others have been demonstrated to have existed in the same location for decades (Lindenbaum et al. 2008). Persistence may be linked to environmental favourability and pressures. Fishing pressures that remove settling cues or required habitat components for larvae may prevent reestablishment of this habitat type.

On sheltered shores and estuaries in southern Europe dense beds of oysters (*Crassostrea gigas*) occur. These tend to form sheets on rocky shores and on artificial substrata such as piles or quaysides. This species is found in southern parts of Britain but is restricted to occasional spat falls rather than established populations. Beds of European oysters (*Ostrea edulis*) used to occur widely in the subtidal regions of northern Europe. To what extent these were natural is unknown; in some locations relaying of oyster beds is thought to date back to Roman times. Many of these beds have been dredged out, or have been heavily impacted by disease (http://www.ukmarinesac.org.uk).

3.2.2 Description of Fishing Activities

The commercial fishing activities likely to occur within these habitat groups are shown in Table 6.

Table 6 Commercial fishing activities associated with bivalve beds

Habitat Group	Shore Access	Hand Gathering	Static Gears	Towed gears
H4	✓	✓		
H4 (subtidal)			✓	✓
H5	✓	✓	✓	✓
H25	✓	✓	✓	✓

The characterising organisms (bivalves) within this habitat group are targeted commercially. As these provide much of the structural habitat complexity, and alter and stabilise sediment, removal of these species can be seen as a physical pressure which has the potential to alter habitat type. In the past, intertidal mussel beds were exploited by hand with a variety of simple hand tools. These artisanal fisheries still persist on a small scale and, in the absence of adequate recruitment, can significantly deplete the biomass on the most accessible beds. Where the beds extend into low water channels, as in the Conwy Estuary, long-handled rakes or long handled tongs may be used to lift clumps from the seabed into a boat. The biggest yields from mussel fisheries in England and Wales now involve a measure of cultivation known as relaying, in which young seed mussels are transplanted onto plots (lays) in the low intertidal or very shallow sublittoral, where they grow well. This movement, and the subsequent harvesting, is now often done with quite large purpose built dredging vessels capable of carrying 12 tonnes of mussels or more. Raft-and-line mussel farms may have impacts on the local benthos (see McKay and Fowler 1997) but do not generally impact directly onto *Mytilus* biogenic reef areas.

Intertidal reefs can be subject to hand gathering although this is generally small scale.

3.2.3 Reviews and Sensitivity Assessments

Information on the impacts of fishing on oyster and mussel beds are provided in Lenihan and Peterson 1998, Magorrian and Service 1998, Cranfield et al. 1999, Hoffmann and Dolmer 2000, Dolmer et al. 2001, and Roberts et al. 2004, whilst Smith and Murray (2005), assessed the impacts of bait collection and trampling on *Mytilus californianus* mussel beds in southern California. Macdonald et al. (1996) reviewed the sensitivity of seabed types and benthic species, including *Modiolus modiolus*, to a ‘single encounter’ with static and towed fishing gears.

The Marine Life Information Network (MarLIN) has produced sensitivity assessments using the MarLIN approach (including factors relevant to fishing activities) for *Mytilus edulis*, other mussel species and native oyster. They have also assessed the sensitivity of a number of habitat types in which mussels are found.

- *Mytilus edulis* and *Fucus vesiculosus* on moderately exposed mid eulittoral rock;
- *Mytilus edulis* and piddocks on eulittoral firm clay;
- *Mytilus edulis* and barnacles on very exposed eulittoral rock;
- *Mytilus edulis* beds with hydroids and ascidians on tide-swept moderately exposed circalittoral rock;
- *Mytilus edulis* beds on reduced salinity tide-swept infralittoral rock;
- *Mytilus edulis* beds in variable salinity infralittoral mixed sediment;
- *Modiolus modiolus* beds with hydroids and red seaweeds on tide-swept circalittoral mixed substrata; and
- *Ostrea edulis beds* on shallow sublittoral muddy sediment

MarLIN assessments of sensitivity (intolerance and recoverability) are summarised in Appendix 1. The sensitivity of intertidal mussel beds to impacts caused by access to fishing grounds was reviewed by Tyler-Walters and Arnold (2008). Hall et al. (2008) assessed the sensitivity of oyster beds to fishing activities.

3.2.4 Resistance (Tolerance)

The characterising species associated with these habitat groups are sessile (fixed to the substratum) as adults (juvenile mussels may drift on byssus threads and undertake long distance migrations). These species are therefore unable to move to avoid direct contact with towed gears.

The characterising species of these habitat groups (i.e. oysters and mussels) have outer shells which provide some protection against physical impacts. However, these shells can be broken by direct pressure from trampling and/or towed gears. *Mytilus* is probably less affected by incidental damage due to fishing for other organisms than other biogenic reef communities. Reise and Schubert (1987) reported that reefs of *S. spinulosa*, lost from areas of the southern North Sea due to shrimp fishing, were replaced by *M. edulis* communities and assemblages of sand dwelling amphipods. The mussels are also now exploited in addition to the shrimps (Holt et al. 1998). Damaged individuals are vulnerable to mobile predators such as crustaceans and starfish.

When fished by hand at moderate levels using traditional methods the biogenic reefs will probably retain most of their intrinsic biodiversity. Many of the same species may even survive in good numbers under cultivation regimes. Natural mussel beds are, however, vulnerable to over-exploitation. Dolmer et al. (2001) showed that mussel dredging caused a short-term decrease in the density and number of species recorded within a dredged area compared to control and boundary areas. These authors also found that brown shrimp (*Crangon crangon*) invaded the dredged area to prey on polychaetes exposed by the dredging.

Biogenic Features

The characteristic species associated with these habitat groups form reefs at the surface which are vulnerable to physical damage from trampling (intertidal) while subtidal reefs may be damaged by the passage of towed gears. Removal of the characterising species alters the habitat complexity and may reduce the diversity of the biological assemblage associated with bivalve reefs.

There is considerable evidence that scallop dredging has been responsible for causing widespread damage to *Modiolus modiolus* beds in Strangford Lough and the Isle of Man (Service and Magorrian 1997; Magorrian and Service 1998; Holt et al. 1998; Hiscock et al. 2005). These effects include the lowering of reef height and loss of associated epifauna especially emergent species such as *Alcyonium digitatum*.

Geomorphological/Sedimentary Features

Mussel dredging may alter the topography of the benthic habitat, remove substrate, resuspend bottom sediment or result in changes in sediment type. For example, Dolmer et al. (2001) observed that mussel dredging resulted in the formation of 2-5cm deep furrows in the seabed, although they did not record any changes in the median diameter of the sediment.

3.2.5 Resilience (Recovery)

Mussels and oysters produce large numbers of planktonic larvae which can potentially recolonise damaged/denuded areas, although larval mortality may be high and hence only a small proportion survive to settle. The main determinants of larval settlement are substratum availability, climatic and hydrodynamic factors and adult abundance.

The bivalve species reviewed in this section are relatively large, long-lived species. The life span of *Ostrea edulis* is 5-10 years, whilst *Mytilus edulis* longevity is variable dependant on locality and habitat but specimens have been reported to reach 18-24 years. *Modiolus modiolus* has a life span of 20-100 years only reaching sexual maturity at 3-8 years. Pre and post settlement mortality of *M. modiolus* larvae is high and recruitment can be sporadic and highly variable. Macdonald et al. (1996) assessed the sensitivity of benthic species to different fishing activities by combining ‘scores’ of the fragility of each species with its ability to recover. *M. modiolus* was assessed as having the lowest recovery ‘score’ related to the fact that it is a slow growing, poorly recruiting species. Hence, under high fishing intensities where bivalve bed removal is widespread, recovery of habitat complexity to its former state may take a number of years.

Biogenic Features

The settlement of *Mytilus* larvae is encouraged by the presence of con-specifics so that reductions in the size of mussel beds through mussel dredging may inhibit re-establishment of a replacement colony.

Geomorphological/Sedimentary Features

Changes in sediment stability, caused by the removal of shell and stone by towed gears or burial of these through siltation (arising from sediment plumes from fishing activities) may inhibit larval settlement and re-establishment of biogenic structures.

3.3 Eelgrass

The habitat types (and corresponding CCW habitat groups and EUNIS classification) referred to in this review section are shown in Table 7.

Table 7 Eelgrass habitat groups

Habitat types	CCW habitat group	EUNIS L3	EUNIS L4	EUNIS Name
Angiosperm-dominant soft littoral sediments, fisheries nursery grounds	H30	A2.6	A2.61	Seagrass beds on littoral sediments
Subtidal Seagrass beds	H30	A5.5	A5.53,	Sublittoral seagrass beds
	H30	A5.5	A5.54	Angiosperm communities in reduced salinity

3.3.1 Brief Description of Habitat Group(s):

Eelgrasses (*Zostera* spp.) are submerged, rooted, grass-like plants. Two species of *Zostera* occur in the UK, namely the common eelgrass *Zostera marina* and dwarf eelgrass *Z. noltii*. In most UK literature another species, *Z. augustilfolia* is regarded as distinct from *Z. marina* (Foden and Brazier 2007). However, *Z. augustilfolia* is currently regarded as a variant of *Z. marina* and not a distinct species (Den Hartog and Kuo 2006).

Eelgrass occupies a wide range of habitats characterized by variations in salinity, wave and current energies, nutrient content of sediments, and substrates that contain various amounts of sand, gravel, rock and mud. However, eelgrass is limited to shallow water habitats due to its dependence on relatively high levels of light.

Eelgrass meadows or beds are recognized as one of the most productive of coastal habitats, providing habitat for a variety of organisms and contributing to the overall productivity of the marine ecosystem. They are a structurally complex habitat that attracts fish and other species. Eelgrass is also an important food for overwintering waterfowl.

Many eelgrass populations are perennial. As the rhizomes grow and extend horizontally through the sediment, new lateral shoots develop and produce clusters of leaves. This form of vegetative reproduction enables eelgrass to spread and form dense meadows that can persist year after year. The plants also reproduce sexually through flowering, pollination, and seed germination. This enables eelgrass to colonize new areas and to reoccur in areas that are subjected to stressors such as fishing.

3.3.2 Description of Fishing Activities

The commercial fishing activities likely to occur within these habitat groups are shown in Table 8.

Table 8 Commercial fishing activities associated with eelgrass habitats

Habitat Group	Shore Access	Hand Gathering	Static Gears	Towed gears
H30	✓	✓		
H30 (subtidal)			✓	✓

Intertidal eelgrass beds may be vulnerable to hand gathering (for other target species such as bivalves) and access to fishing grounds. Subtidal eelgrass beds may be subject to static gear placement, anchoring and the use of towed gears to capture target species.

3.3.3 Reviews and Sensitivity Assessments

The sensitivity of UK *Zostera* beds to anthropogenic changes was assessed by Holt et al. (1997), Davison and Hughes (1998) and MarLIN. Extensive reviews of the effects of commercial and fishing activities on eelgrass have been produced by the Department of Environment Protection, Connecticut (McCarthy and Preslli 2007) and the Atlantic States Marine Fisheries Commission (ASMFC 2000). A useful overview of European seagrass and management is provided by Borum et al. (2004).

Macdonald et al. (1996) reviewed the sensitivity of seabed types and benthic species, including *Zostera marina*, to a ‘single encounter’ with static and towed fishing gears. McMath et al. (2000) provided guidance on the recovery ability of marine species, including *Zostera marina*, following maritime activities, based on each species life history traits (recruitment, recolonization and regeneration).The sensitivity of intertidal eelgrass to impacts from foot and vehicle access to fishing grounds was assessed by Tyler-Walters et al. (2008). Hall et al. (2008) assessed eelgrass sensitivity to fishing impacts using the modified ‘Beaumaris approach’.

MarLIN has produced a sensitivity assessment (using the MarLIN approach which includes factors relevant to fishing activities) for the species *Zostera marina* and *Z. noltii*, and the biotopes:

- *Zostera noltii* beds in littoral muddy sand;
- *Zostera marina/angustifolia* in infralittoral muddy sands.

MarLIN assessments of sensitivity (intolerance and recoverability) are summarised in Appendix 1.

3.3.4 Resistance (Tolerance)

Intertidal *Zostera* beds are relatively fragile and therefore sensitive to trampling (S.J. Hawkins, pers. comm. cited in Holt et al. 1997; Thom 1993). Tyler-Walters and Arnold (2008) reviewed the literature on the effects of trampling and vehicle access on intertidal eelgrass and concluded that these were detrimental to beds, resulting in the loss of biomass and standing crop.

Subtidally, deployment of towed and static gears can result in physical damage to the above surface parts of the plants and the root systems, which are found within the top 20cm of sediment. Leaf shearing results when leaves are cut and repeated occurrence may cause plant death where most of the plant resources must be directed to leaf replacement. Seed or flower shearing can prevent sexual reproduction.

Otter trawling in beds of the seagrass *Posidonia oceanica* have been shown to cause significant damage to plants, leading to the loss of entire meadows (Ardizzone and Pelusi 1983, Ardizzone et al. 2000). The steel otter doors, ground cable, and net lead to sediment scouring, plant damage, up-rooting and sub-surface rhizome damage. Trawling near eelgrass beds will result in sediment disturbance and suspension reducing light penetration and the burial/smothering of eelgrass.

Increased dredging for scallops in North Carolina led to significantly reduced levels of eelgrass (*Z. marina*) biomass and shoot number ($P < 0.01$) on both hard sand and soft mud bottoms (Fonseca et al. 1984).

Zostera is very sensitive to hydraulic bivalve fishing due to damage and break up of rhizomes. Davison & Hughes (1998) reported that damage from mechanical dredging of cockles to intertidal *Zostera* beds in the Solway Firth resulted in closure of the fishery, while Eno et al. (1997) reported that *Zostera* beds were damaged by dredging for *Mercenaria*. Dislodged rhizomes cannot re-root.

Based on observations and the general physical characteristics of lobster pots, it is considered that pots consistently set and hauled in an eelgrass meadow can cause damage by leaf shearing, damaging meristems, uprooting plants and, if left long enough on the bottom, can cause damage by smothering and light attenuation. The extent of damage by pots depends on the number of pots set, the set over period and hauling frequency (ASMFC 2000).

The type and extent of damage that moorings can cause has been extensively studied in Australia, which has a number of seagrass species (see Hastings et al. 1995). Setting and retrieving anchors in eelgrass meadows can dislodge and damage eelgrass leaves and rhizomes. The anchor chain can damage plants in numerous ways, ranging from leaf shearing to below ground impacts. In cases where a single mooring is used, the mooring chain is dragged across the bottom repeatedly with each tidal cycle and changes in wind direction. With repeated scouring the chain completely denuded a circular area defined by the length of the chain and angle of sweep. A boat that swings around the mooring will form a circular scar in the eelgrass meadow (McCarthy and Preslli 2007).

Biogenic Features

Eelgrass beds create structurally complex habitats; damage and removal of plants will reduce the structural complexity of this habitat.

Geomorphological /Sedimentary Features

When sediments are dislodged or resuspended eelgrass plants are likely to be buried or smothered. Plants may be killed depending on the extent and duration of smothering. Changes in bed extent and fragmentation may lead to further losses of beds through sediment loss and destabilisation (Holt et al. 1997).

3.3.5 Resilience (Recovery)

Intertidal populations of *Z. angustifolia* have been described as annual (Cleator 1993). The rhizomes of *Z. marina* are described as long-lived (Holt et al. 1997) so that beds can, potentially, be very old.

Eelgrass beds are reported to undergo very strong seasonal variation. Intertidal populations are often annual and can undergo complete dieback in winter with recovery dependent on local seed supply (Holt et al. 1997). In perennial populations die back of above ground parts are less significant and recovery is through vegetative growth. Eelgrass beds are also spatially dynamic, with advancing and leading edges causing changes in coverage, the beds expand either through vegetative growth from shooting rhizomes that have survived the winter, or sexually, by production of seed. Subtidal *Z. marina* beds in the UK are perennial and are believed to persist almost completely as a result of vegetative growth rather than by seed. Growth of individual plants occurs during the spring and summer.

Recovery rates will therefore depend on supply of rhizomes or seeds. Given that fragmentation of beds can cause further losses, recovery may be slow, particularly in subtidal areas. However, Dean et al. (1998 cited McMath et al. 2000) reported *Zostera* species can quickly regenerate lost blades.

Scars may remain un-vegetated for a number of years. Studies in Florida have estimated that scars typically require from three to seven years to re-vegetate, and possibly longer in severe cases involving very deep propeller scars and vessel groundings. In some cases scars expand and coalesce to form larger denuded areas. Re-vegetation rates, as well as the potential for scar expansion, depend upon a number of factors, including the species of seagrass, sediment characteristics, bathymetry and the prevailing wind and current patterns (McCarthy and Preslli 2007).

Neckles et al. (2005) found that substantial differences in eelgrass biomass persisted for up to seven years after towed gears were used to harvest *Mytilus edulis* and they predicted that approximately 10 years would be required for the most intensely disturbed areas to recover although this may require 20 years in areas where habitat suitability was lower.

The dispersal range of seagrass seeds is a very poorly studied aspect of their reproductive ecology, and robust estimates of dispersal events are only available for *Zostera marina* populations, for which 95 per cent of the seeds are retained within 30 m from the source.

McMath et al. (2000) calculated the likely recovery potential of *Zostera marina* to human maritime activities, based on the recruitment, recolonization and regenerative characteristics of the species. On a scale of 1-100 (where 1 represented excellent recovery following disturbance and 100 represented no species recovery), the authors calculated that *Z. marina* had an intermediate recovery score of 49.

Biogenic Features

Eelgrass beds create structurally complex habitats, recovery of this feature from fishing impacts will result in the restoration of a more structurally complex habitat.

Geomorphological/Sedimentary Features

Eelgrass beds form in areas of soft sediment and recovery of disturbed areas, such as the infilling of scour pits, is predicted to occur through natural sediment movements. The rate of this recovery will depend on local hydrographic and sediment characteristics such as sediment type and water flow rates.

Eelgrass plants enhance sedimentation by slowing water currents and trapping sediment particles, hence these processes will depend on re-establishment of the bed.

3.4 Faunal Turfs

The habitat types (and corresponding CCW habitat groups and EUNIS classification) referred to in this review section are shown in Table 9.

Table 9 Faunal turf habitat groups

Habitat types	**CCW habitat group**	**EUNIS L3**	**EUNIS L4**	**EUNIS Name**
Predominantly subtidal rock with low-lying and fast growing faunal turf	H20	A3.1	A3.14	Encrusting algal communities
	H20	A3.7	A3.71	Robust faunal cushions and crusts in surge gullies and caves
	H20	A4.1	A4.11	Very tide-swept faunal communities on circalittoral rock
	H20	A4.1	A4.13	Mixed faunal turf communities on circalittoral rock
	H20	A4.2	A4.21	Echinoderms and crustose communities on circalittoral rock
	H20	A4.2	A4.25	Circalittoral faunal communities in variable salinity
	H20	A4.7	A4.71	Communities of circalittoral caves and overhangs
	H20	A4.7	A4.72	Circalittoral fouling faunal communities

3.4.1 Brief Description of Habitat Group(s):

'Faunal turfs' are assemblages of attached animals growing on hard substrata. These organisms can vary substantially in growth form, ranging from low (< 1cm height) encrusting forms (e.g. many ectoprocts (sea mats) and sponges), to tall erect forms such as alcyonarians (soft corals) and gorgonians (sea fans), which may exceed 25 cm in height. This section focuses on fishing impacts on low-lying and fast growing faunal turfs. Impacts on larger, long-lived species are discussed separately in Section 3.10.

Circalittoral faunal turf (CFT) occurs wherever rock substratum extends substantially below extreme low water at spring tide (ELWS), and will be best developed where the rock extends to well below the depths supporting profuse algal growth (10 m or more). The species diversity of CFT communities is generally high and also supports shellfisheries of considerable economic importance.

Echinoderms and crustose communities on circalittoral rock occur on wave-exposed, moderately strong to weakly tide-swept, circalittoral bedrock and boulders. Echinoderms, faunal (*Parasmittina trispinosa*) and algal crusts (red encrusting algae) dominate this biotope.

3.4.2 Description of Fishing Activities

The commercial fishing activities likely to occur within these habitat groups are shown in Table 10.

Table 10 Commercial fishing activities associated with faunal turfs

Habitat Group	Shore Access	Hand Gathering	Static Gears	Towed gears
H20			✓	✓

Static gear is deployed regularly on rocky grounds, either in the form of pots or creels, or as bottom set gill or trammel nets. Whilst the potential for damage is lower per unit deployment compared to towed gear, there is a risk of cumulative damage to sensitive species if use is intensive. Damage could be caused during the setting of pots or nets and their associated ground lines and anchors and by their movement over the bottom during rough weather and during recovery.

Towed gear is not generally a major threat to most faunal turf biotopes since the generally steep and rocky substrata are unsuitable for both trawls and dredges. However there are types of towed gear designed for rocky areas - the rock-hopper otter trawl, and the Newhaven scallop dredge - and these could pose a risk to faunal turf communities on gently sloping or level rock.

3.4.3 Reviews and Sensitivity Assessments

The vast majority of studies investigating the impact of fishing activity and gears on benthic communities have focussed on the impacts on the benthic communities of soft-sediments such as sand. As such, there is generally less information available covering the impacts on other habitat groups such as the one under review in this section. The sensitivity of circalittoral faunal turf biotopes to human activities including fishing was reviewed by Hartnoll (1998), whilst the sensitivity of various species and benthic communities on subtidal rock habitats, including fast growing and low-lying faunal turfs, was reviewed by MacDonald et al. (1996) and Hall et al. (2008).

MarLIN has produced a sensitivity assessment (using the MarLIN approach which includes factors relevant to fishing activities) for the following relevant biotopes.

- *Bugula* spp. and other bryozoans on vertical moderately exposed circalittoral rock.
- *Alcyonium digitatum* with a bryozoan, hydroid and ascidian turf on moderately exposed vertical infralittoral rock.
- Sponge crusts and anemones on wave-surged vertical infralittoral rock.

- Erect sponges, *Eunicella verrucosa* and *Pentapora fascialis* on slightly tide-swept moderately exposed circalittoral rock.
- *Flustra foliacea* and other hydroid/bryozoan turf species on slightly scoured circalittoral rock or mixed substrata.
- *Molgula manhattensis* and *Polycarpa* spp. with erect sponges on tide-swept moderately exposed circalittoral rock.
- Faunal and algal crusts, *Echinus esculentus*, sparse *Alcyonium digitatum* and grazing-tolerant fauna on moderately exposed circalittoral rock.

MarLIN assessments of sensitivity (intolerance and recoverability) are summarised in Appendix 1.

3.4.4 Resistance (Tolerance)

In general, encrusting, sessile epifauna are known to be vulnerable to removal and damage by towed gears (Engel and Kvitek 1998, McConnaughey et al. 2000; Eleftheriou and Robertson 1992). The deployment of towed gears designed for rocky habitats, for example rock-hopper trawls, could potentially damage and entangle epifauna in these habitats, especially fragile long-lived species (Hartnoll, 1998; see also Section 3.10). Indirectly, towed gears could impact faunal turf communities through increasing suspended sediment load in the water and subsequent siltation (Hartnoll, 1998).

Compared to towed gear, benthic communities are relatively unaffected by static fishing gears (pots, long-lines or anchored nets) due to the relatively small area of seabed directly affected (Kinnear et al. 1996; Jennings and Kaiser 1998; Eno et al. 2001). The ability of epifauna to resist impacts from static gears varies between species (see Section 3.10) and the degree of impact will depend on the intensity of the fishing and the duration. Weights and ropes associated with static gears also have the potential to damage epifaunal species.

Biogenic Features

Demersal towed gear damage colonial epifaunal taxa (e.g. algae, sponges, corals, colonial tube worms, hydroids, bryozoans), which provide a three-dimensional habitat for other animals (Jennings and Kaiser 1998; Hall 1999).

Geomorphological/Sedimentary Features

Hard substrates are relatively resistant to physical damage from fishing gears. However, towed gears may damage softer rock types or may physically damage the substrate, reducing complexity.

3.4.5 Resilience (Recovery)

The recovery of the assemblage will depend on the life-history characteristics of the species affected, including the ability of damaged adults to repair/regenerate lost or damaged parts and/or the ability of larvae to reach and recolonise the habitat (i.e. recovery will depend on the species recruitment and/or growth rate; MacDonald et al 1996). In an assessment of benthic species sensitivity to fishing disturbance, MacDonald et al. (1996) classified the species *Nemertesia antennina* and *Antedon bifida*, which may be considered as components of faunal turf, as having a ‘short’ recovery potential, whilst *Flustra foliacea* was classified as having a ‘moderate’ recovery potential.

Biogenic Features

Much of the structural complexity of this habitat type is provided by the faunal turf, such that recovery of the degree of biogenic structure/complexity is dependent on recovery of these components.

Geomorphological/Sedimentary Features

Hard substrata are relatively resistant to physical damage from fishing gears. However, towed gears may damage softer rock types or may physically damage the substrate, reducing complexity, with low levels, if any, of recoverability from this impact.

3.5 Macroalgae dominated subtidal hard substrata

The habitat types (and corresponding CCW habitat groups and EUNIS classification) referred to in this review section are shown in Table 11.

3.5.1 Brief Description of Habitat Group(s):

Macroalgae exist in two taxonomic Kingdoms, the plants (Plantae) and the chromists (Chromista) (Appeltans et al. 2010). Nevertheless, they fall into three broad groups; the green algae (Chlorophyta, Plantae), red algae (Rhodophyta, Plantae), and the brown and yellow-green algae (Ochrophyta, Chromista). These groups have different environmental tolerances; red algae for example are the most tolerant of reduced light conditions. Red and brown algae require hard substrates for attachment.

The habitat types considered in this review all include kelps (brown algae). The kelp species in the UK are most frequently found attached to submerged bedrock. However, given adequate water movement in the form of tidal currents rather than wave action, large kelp plants may frequently be found attached to cobbles and pebbles. Kelps are found in almost all locations where some form of hard substratum is available within the euphotic zone in UK waters.

Table 11 Macroalgae dominated hard substrata habitat groups

Habitat types	CCW habitat group	EUNIS L3	EUNIS L4	EUNIS Name
Shallow subtidal rock with kelp	H22	A3.1	A3.11	Kelp with cushion fauna and/or foliose red seaweeds
	H22	A3.2	A3.21	Kelp and red seaweeds (moderate energy infralittoral rock)
	H22	A3.3	A3.31	Silted kelp on low energy infralittoral rock with full salinity
Kelp and seaweed communities on sand scoured rock	H23	A3.1	A3.12	Sediment-affected or disturbed kelp and seaweed communities
	H23	A5.5	A5.52	Kelp and seaweed communities on sublittoral sediment

Water movement and the hydrographic regime have a number of effects on both individual kelp plants and on the kelp bed as a whole. For example, in areas where the kelp bed is exposed to heavy wave action (e.g. on an open coast, a headland or at the mouth of a loch), the plants and animals found differ markedly from areas where there is little wave action (within a bay or cove or within a loch).

The faunal and floral diversity of kelp biotopes is extremely rich which is in part associated with the diversity of available food sources but is also due to the physical and structural diversity within the biotopes, with the many and various exploitable niches available.

Within kelp beds, some fauna may be mainly or entirely restricted to the kelp plants themselves, e.g. species found in the kelp holdfasts, although much of the rock crevice fauna or sediment infauna may occur more or less independently of the presence of kelp. The flora found in kelp beds may also not be restricted to this habitat but the complex interactions of the grazing species found in kelp beds and the several habitats available for colonization within the kelp bed may lead to a wide diversity of seaweeds being present within a given area.

Species Groups

Shallow subtidal rocky habitats exposed to extreme wave action or strong tidal streams, typically support a community of the kelp *Laminaria hyperborea* with foliose seaweeds and animals, the latter tending to become more prominent in areas of strongest water movement. In some areas, there may be a band of dense foliose seaweeds (reds or browns) below the main kelp zone. The sublittoral fringe is characterised by dabberlocks *Alaria esculenta*.

Similar habitats subject to moderate wave exposure, or moderately strong tidal streams on more sheltered coasts, typically support a narrow band of the kelp *Laminaria digitata*, with associated seaweed communities (predominantly red algae with a variety of filamentous species) in the sublittoral fringe, which lies above a *Laminaria hyperborea* forest and park.

Shallow subtidal rocky habitats in wave and tide-sheltered areas support silty communities with *Laminaria hyperborea* and/or *Laminaria saccharina,* and associated seaweeds which are typically silt-tolerant and include a high proportion of delicate filamentous types. In these areas, the lower infralittoral zone is subject to intense grazing by urchins and chitons and may have poorly developed seaweed communities.

Where the habitat is subject to disturbance through mobility of the substratum (boulders or cobbles) or abrasion/covering by nearby coarse sediments or suspended particulate matter (sand), the associated communities are variable in character. The typical *Laminaria hyperborea* and red seaweed communities of stable open coast rocky habitats are replaced by those which include species that are more ephemeral or those tolerant of sand and gravel abrasion. *Laminaria saccharina*, *Saccorhiza polyschides* or *Halidrys siliquosa* may be prominent components of the community.

Where kelp and seaweed occur on sublittoral sediment in sheltered habitats, communities typically include the kelp *Laminaria saccharina*, the bootlace weed *Chorda filum* and various red and brown seaweeds, particularly filamentous types, growing on shells/small stones on the sediment surface or developing as loose-lying mats on the sediment surface.

3.5.2 Description of Fishing Activities

The commercial fishing activities likely to occur within these habitat groups are shown in Table 12.

Table 12 Commercial fishing activities associated macroalgae dominated hard substrata.

Habitat Group	Shore Access	Hand Gathering	Static Gears	Towed gears
H22			✓	✓
H23			✓	✓

The habitat groups considered here are subtidal so shore access and trampling are not relevant to this section.

Generally, steep and rocky substrata are unsuitable for both trawls and dredges. However there are types of towed gear designed for rocky areas - the rock-hopper otter trawl, and the Newhaven scallop dredge - and these could pose a risk to communities on gently sloping or level rock, or on mixed substrata.

Static gear is deployed regularly on shallow rocky grounds, either in the form of pots or creels, or as bottom set gill or trammel nets. Whilst the potential for damage is lower per unit deployment compared to towed gear, there is a risk of cumulative damage to sensitive species if use is intensive. Damage could be caused during the setting of pots or nets and their associated ground lines and anchors, and by their movement over the bottom during rough weather and during recovery.

Algae, including kelp, are harvested for consumption, alginate production, use in beauty and health products and land fertiliser, although commercial mechanical harvesting has not occurred in the UK (Birkett et al. 1998b). Mechanical harvesting of kelp, using kelp dredgers, has been ‘trialled’ in Scotland[1].

3.5.3 Reviews and Sensitivity Assessments

Studies of the impacts of fishing activities on benthic communities have predominantly focussed on soft-sediments, such as sand and mud, and there is a relative paucity of

[1] http://www.snh.org.uk/publications/on-line/livinglandscapes/kelp/harvesting.asp

information regarding the impacts of fishing activities on this habitat group. The sensitivity of some macroalgae species to fishing disturbance, including the ‘recovery’ potential of these species, was assessed by Macdonald et al. (1996) and McMath et al. (2000).

MarLIN has produced sensitivity assessments using the MarLIN approach (including factors relevant to fishing activities) for *Laminaria hyperborea, L. digitata, Alaria esculenta, Saccorhiza polyschides and Halidrys siliquosa*. They have also assessed the sensitivity of a number of habitat types in which macroalgae is found. MarLIN assessments of sensitivity (intolerance and recoverability) are summarised in Appendix 1.

3.5.4 Resistance (Tolerance)

It is likely that kelp would be removed / damaged by towed gears. Similarly, it is likely that static gears, and weights and ropes associated with such gears, would have the potential to damage or remove kelp through entanglement.

Biogenic Features

Kelp and other macroalgae contribute to the structural complexity of this habitat type and therefore the removal of these through fishing activities would reduce habitat complexity.

Geomorphological/Sedimentary Features

Hard substrates are relatively resistant to physical damage from fishing gears. However, towed gears may damage softer rock types or may physically damage the substrate, reducing complexity.

3.5.5 Resilience (Recovery)

Recovery will be dependent upon the life history characteristics of the species affected. In an assessment of benthic species sensitivity to fishing disturbance, Macdonald et al. (1996) classified the kelp species *Laminaria hyperborea* (mature) as having ‘moderate’ recovery potential. Using a similar methodology, McMath et al. (2000) scored the recruitment ability of kelp as 1-20 (on a scale of 1-100, where ‘1’ represents the maximum recruitment success and 100 represents no recruitment ability) based on life history characteristics (rapid growth rates of 1-5cm/week, sexual maturity at 1-2years and frequent reproduction). The regenerative ability of kelp was ‘scored’ as 20-30 (out of a scale of 1-100, where ‘1’ represents the maximum regeneration ability and 100 represents no regeneration ability) as rapid re-growth of kelp blades can occur following damage/removal, providing the meristematic basal area of the blade remains intact.

Biogenic Features

Much of the structural complexity of this habitat type is provided by macroalgae, so that recovery of the degree of biogenic structure/complexity is dependent on recovery of these components.

Geomorphological/Sedimentary Features

Hard substrata are predicted to have low recoverability from physical damage.

3.6 Mixed Sediments

The habitat types (and corresponding CCW habitat groups and EUNIS classification) referred to in this review section are shown in Table 13.

Table 13 Mixed Sediment habitat groups

Habitat types	CCW habitat group	EUNIS L3	EUNIS L4	EUNIS Name
Stable, species rich mixed sediments	H28	A5.4	A5.42	Sublittoral mixed sediment in variable salinity (estuaries)
	H28	A5.4	A5.43	Infralittoral mixed sediments
	H28	A5.4	A5.44	Circalittoral mixed sediments
	H28	A5.4	A5.45	Deep circalittoral mixed sediments
Coarse sands and gravels with communities characterised by burrowing large/long lived bivalves	H16	A5.1	A5.12	Sublittoral coarse sediment in variable salinity (estuaries)
	H16	A5.1	A5.13	Infralittoral coarse sediment
Unstable cobbles, pebbles, gravels and/or coarse sands supporting relatively robust communities	H29	A5.1	A5.12	Sublittoral coarse sediment in variable salinity (estuaries)
	H29	A5.1	A5.14	Circalittoral coarse sediment

3.6.1 Brief Description of Habitat Group(s):

These habitats are all dominated by infaunal assemblages but also support a range of epibenthic organisms (Hall et al. 2008). This habitat group covers a wide variety of habitats and associated fauna and these different habitats will vary in their sensitivity to fishing impacts.

Sublittoral sand and gravel habitats occur in a wide variety of environments, from sheltered (sea lochs, enclosed bays and estuaries) to highly exposed conditions (open coast). The particle structure of these habitats ranges from mainly sand, through various combinations of sand and gravel, to mainly gravel. While very large areas of seabed are covered by sand and gravel in various mixes, much of this area is covered by only very thin deposits over bedrock, glacial drift or mud. The strength of tidal currents and exposure to wave action are important determinants of the topography and stability of sand and gravel habitats.

Sand and gravel habitats that are exposed to variable salinity in the mid- and upper regions of estuaries, and those exposed to strong tidal currents or wave action, have a low diversity. They are inhabited by robust, errant fauna specific to the habitat such as small polychaetes, small or rapidly burrowing bivalves and amphipods. The epifauna in these habitats tends to be dominated by mobile predatory species.

Sand mixed with cobbles and pebbles that is exposed to strong tidal streams and sand scour is characterised by conspicuous hydroids and bryozoans. These fauna increase the structural complexity of this habitat and may provide an important microhabitat for smaller fauna such as amphipods and shrimps.

Mixed sediment habitats that are less perturbed by natural disturbance are among the most diverse marine habitats with a wide range of anemones, polychaetes, bivalves, amphipods and both mobile and sessile epifauna.

Circalittoral gravels, sands and shell gravel are dominated by thick-shelled bivalve and echinoderms species, (e.g. *Pecten maximus, Circomphalus casina, Ensis arcuatus* and *Clausinella fasciata*), sessile sea cucumbers (*Neopentadactyla mixta*), and sea urchins (*Psammechinus miliaris* and *Spatangus purpureus*). These biotopes have been described by previous workers as the 'Boreal Off-Shore Gravel Association' and the 'Deep Venus Community' and can be found in Shetland, the western coasts, Irish Sea and English Channel.

Gravel habitats are functionally important as they provide spawning substrate for the eggs of some demersal fish species and act as nursery grounds for other fish species (e.g. juvenile cod and haddock in Georges Bank; northwest Atlantic, Collie et al. 2005).

3.6.2 Description of Fishing Activities

The commercial fishing activities likely to occur within these habitat groups are shown in Table 14.

Table 14 Commercial fishing activities associated with mixed sediment habitats

Habitat Group	Shore Access	Hand Gathering	Static Gears	Towed gears
H28			✓	✓
H16			✓	✓
H29			✓	✓

This habitat group is not exposed to impacts from hand gathering or shore access to fishing grounds as it occurs subtidally. The group is subject to fishing impacts from towed and static gears. Gravel substrata are fished with scallop dredges (particularly in the English Channel and northern Irish Sea in the UK).

3.6.3 Reviews and Sensitivity Assessments

Numerous studies have assessed the impact and recovery of benthic communities to towed demersal gears on the mixed sediment habitats described above, including Freese et al. (1999), Collie et al. (1997, 2005, 2009), Kaiser et al. (2000), Bradshaw et al. (2000, 2001, 2002) and Veale et al. (2000). Reviews of the impacts of different fishing gears on these benthic habitat groups include: Hall et al. (2008), Kaiser et al. (2002, 2006), Johnson (2002) and Thrush and Dayton (2002). Studies looking at the impact of static gears in various habitats including mixed sediments include Blyth et al (2004) and Eno et al. (2001).

The relative sensitivity of different seabed types (including clean and mixed grounds) and associated benthic species to fishing disturbance was assessed by MacDonald et al. (1996). More recently, the sensitivity of these habitat groups to fishing impacts was assessed by Hall et al. (2008) using the adapted Beaumaris approach.

The Marine Life Information Network (MarLIN) has produced sensitivity assessments using the MarLIN approach (including factors relevant to fishing activities) for mixed sediments, or species found in mixed sediments, including;

- Venerid bivalves in circalittoral coarse sand or gravel
- *Abra alba*, *Nucula nitida* and *Corbula gibba* in circalittoral muddy sand or slightly mixed sediment
- *Halcampa chrysanthellum* and *Edwardsia timida* on sublittoral clean stone gravel

MarLIN assessments of sensitivity (intolerance and recoverability) are summarised in Appendix 1.

3.6.4 Resistance (Tolerance)

The impacts of bottom fishing vary between sediment types and the resident fauna. Impacts of towed demersal gears on gravel are reviewed in Thrush and Dayton (2002) and Johnson (2002). The biological effects of towed gears in this habitat group included:

- reduced numbers of organisms, reduced biomass, and lower species diversity on gravel pavement substratum (Collie et al., 1997);
- the removal and damage of large epifauna (e.g. anemones, sea whips and sponges) on pebble/cobble/boulder substratum (Freese et al.,1999); and
- the loss of sessile, emergent, high biomass species accompanied by an increase in small-bodied infauna less susceptible to physical disturbance on gravel/sand (Kaiser et al, 2000).

Kaiser et al. (2006) undertook a meta-analysis to examine the response and recovery of benthic biota in different habitats to different fishing gears (scallop dredging, otter trawls, beam trawls, intertidal raking and intertidal dredging). The initial and long term impacts of different types of fishing gear are strongly habitat-specific as some habitats are pre-adapted to natural disturbance and are characterised by species that are relatively resistant or can recover rapidly. Gravel habitats were negatively affected in the short and long term by some fishing activities (scallop dredging). However, the initial impacts were of a lesser magnitude compared to that in other less stable habitats (Kaiser et al. 2006).

Benthic communities are relatively unaffected by static fishing gears (fish or crustacean pots, long-lines or anchored nets) compared to towed gears, because of the relatively small area of seabed directly affected (Kinnear et al. 1996; Jennings and Kaiser 1998; Eno et al. 2001). Benthic community biomass in areas subjected to only static gear use has been reported to be significantly greater compared to areas in which trawling has occurred within the last two years (Blyth et al. 2004). However, epifauna may be damaged by weights and ropes or entangled and removed. Where the gear drags or bounces the damage will be more widespread.

Biogenic Features

Trawling activity in gravel habitats can result in decreased biogenic structure by the removal of structure-forming epifauna (Collie et al. 1997; Auster et al. 1996). For example, unfished gravel habitat in Georges Bank, northwest Atlantic, were characterised by bushy epifaunal taxa (bryozoans, hydroids, worm tubes) that provided a complex habitat for shrimps, polychaetes, brittle stars, mussels and small crabs. In areas subjected to high levels of bottom fishing, the epifauna was removed, resulting in reduced habitat complexity and a benthic assemblage dominated by larger, hard-shelled molluscs, scavenging crabs and echinoderms (Collie et al. 1997).

Geomorphological/Sedimentary Features

Impacts of trawling on mixed sediment habitats include tracks in sediment (1-8 cm deep in less compact substrate; Freese et al., 1999), removal of fine sediment, sediment resuspension, smoothing of seafloor and displaced/overturned gravel, stones and boulders (e.g. Auster et al. 1996; Bridger 1972, Engel and Kvitek 1998, Freese et al. 1999, all cited in Johnson 2002).

3.6.5 Resilience (Recovery)

Population recovery rates will be species specific; species such as long-lived bivalves are likely to have long recovery periods from disturbance whilst other populations are likely to recover more rapidly. Megafaunal species (e.g. molluscs, shrimps over 10mm), and especially emergent and sessile species, are generally more vulnerable to fishing effects than macrofaunal species as they are slow growing and take a long time to recuperate from disturbance/harvesting.

The rate of natural disturbance experienced by the habitat will influence recovery rates. In locations subject to high levels of natural disturbance, the biological assemblage will be characterised by species able to withstand and recover from perturbations. Habitats within more stable environments, characterised by high diversity and epifauna, are likely to take longer to recover.

Monitoring of a ‘closed area’ of gravel habitat on Georges Bank, showed that five years after closure of the area to high levels of scallop fishing, the biomass and abundances of certain taxa (including crabs, molluscs, polychaetes and echinoderms) were still increasing (Collie et al. 2005). As such, the authors predicted that the recovery time for gravel habitats was in the order of ten years. Similar recovery rates were observed during 10 years of monitoring of a gravelly habitat off the Isle of Man following closure to scallop dredging (Bradshaw et al. 2000)

Biogenic Features

The populations of sessile epifauna, which provide the biogenic habitat complexity in this habitat group, may recover only slowly from physical damage and disturbance. Recovery rates will be partly determined by species life history traits, fast growing faunal turfs would be predicted to recover faster than slow-growing species (see Sections 3.4 and 3.10). A study by Collie et al. (2009) on northern Georges Bank has shown that the recolonization of defaunated gravel was more rapid for free living species than for structure-forming epifauna. The authors speculate that the slow rate of recolonization of gravel habitat by structure-forming epifauna (sponges, bryozoans, anemones, hydroids, colonial tube worms) following fishing disturbance may be due to factors such as the low survival of recruits of these species, due to intermittent burial of the gravel by migrating sands, and the presence of high numbers of scavengers (crabs, echinoderms, nudibranchs, gastropods), the abundance of which increased rapidly on the gravel post disturbance. Hence, this suggests that the recovery of these habitats may be slower than life history traits predict.

Geomorphological/Sedimentary Features

Dredge marks from towed gear have been shown to be relatively short lived, lasting from a few days to no more than a year in coarse sediments (De Groot and Lindeboom 1994, Lindeboom and de Groot 1998).

3.7 Mud Sediments

The habitat types (and corresponding CCW habitat groups and EUNIS classification) referred to in this review section are shown in Table 15.

Table 15 Mud sediment habitat groups

Habitat types	CCW habitat group	EUNIS L3	EUNIS L4	EUNIS Name
Intertidal and sublittoral sand, muddy gravel	H11	A2.2	A2.24	Polychaete/bivalve dominated muddy sand shores
Intertidal and sublittoral sand, muddy gravel	H11, Subtidal	A2.4	A2.41	*Hediste diversicolor* dominated gravely sandy mud shores
	H11, Subtidal	A5.2	A5.26	Circalittoral muddy sand
Intertidal mud flats, cockle beds (exploited),	H10, H11, H12	A2.3	A2.31	Polychaete/bivalve-dominated mid estuarine mud shores
Intertidal mud flats, cockle beds (exploited),	H10, H11, H13	A2.3	A2.33	Marine mud shores
Intertidal mud flats, cockle beds (exploited),	H10, H11,	A2.4	A2.41	*Hediste diversicolor* dominated gravely sandy mud shores
Subtidal stable muddy sands, sandy muds and muds	H19	A5.2	A5.24	Infralittoral muddy sand
	H19	A5.2	A5.26	Circalittoral muddy sand
	H19	A5.2	A5.27	Deep circalittoral sand
	H19	A5.3	A5.31	Sublittoral mud in low or reduced salinity
	H19	A5.3	A5.33	Infralittoral sandy mud
	H19	A5.3	A5.34	Infralittoral fine mud
	H19	A5.3	A5.35	Circalittoral sandy mud
	H19	A5.3	A5.36	Circalittoral fine mud

3.7.1 Brief Description of Habitat Group(s):

Intertidal mudflats occur predominantly in estuaries and the adjacent sedimentary coastal areas, in sheltered marine bays and semi-enclosed areas including lagoons. Intertidal mudflats are predominantly clay, silt and to a lesser extent very fine sand.

The importance of these habitats centres on their role in the biological and physical functioning of the ecosystems. For example, mudflats produce material for predators, such as birds, fishes and mobile epibenthic invertebrates, and mud and sandflats for coastal protection.

The benthic communities within intertidal sand and mudflats and subtidal sandbanks usually comprise a restricted set of species, which differ according to substrata and environmental variables (described further below). Macrophyte communities are usually poor unless some stones/shells are present for species to attach to. Seagrasses occur in sheltered sand and mudflats both intertidally and in the shallow subtidal (reviewed separately in Section 3.3) whilst in sheltered brackish conditions on the upper shore saltmarsh plants may become established (reviewed separately in Section 3.8).

Intertidal Mudflats

Estuarine mud flats (low energy areas) have well-defined macrobenthic communities similar to the boreal shallow mud community described by Jones (Jones 1950). Often the fauna shows low species diversity, even though biomass may be high but this depends on the amount of silt present. In fully marine areas the organic content is lower and surface deposit-feeding terebellids, spionid polychaetes and the filter-feeding bivalve *Cochlodesma* are common. Firm muds may support piddocks and the boring spionid worm *Polydora ciliata,* while less well-consolidated muds are characterised by other nereid, spionid and capitellid worms.

Mobile Epibenthos

Epifaunal organisms associated with these biotope complexes are predominantly mobile predatory species such as crabs (e.g. *Carcinus maenus*) and shrimps (e.g. *Crangon crangon*) which prey on infaunal populations of small bivalves, polychaetes and crustaceans. Organisms associated with silty sands are predominantly mobile species e.g. the crab *Liocarcinus depurator*.

Subtidal Stable Muddy Sands, Sandy Muds and Muds

Muddy sand comprises of non-cohesive muddy sand (with 5 per cent to 20 per cent silt/clay). Both infralittoral muddy sands (from lower shore down to about 15-20 m) and circalittoral muddy sands (generally in water over 15-20 m deep) support a variety of animal-dominated communities, particularly polychaetes, bivalves and echinoderms (with different species of these taxa characterising the infralittoral and circalittoral zones). The circalittoral habitats tend to be more stable than their infralittoral counterparts and as such support a richer infaunal community.

Sandy mud typically contains over 20 per cent of silt/clay. Infralittoral sandy mud is generally found in sheltered bays or marine inlets and along sheltered areas of open coast. Typical species include a rich variety of polychaetes, tube building amphipods (*Ampelisca* spp.) and deposit feeding bivalves. In circalittoral sandy mud, sea pens (e.g. *Virgularia mirabilis*) and brittlestars (e.g. *Amphiura* spp.) are particularly characteristic of this habitat whilst infaunal species include the tube building polychaetes and deposit feeding bivalves (e.g. *Abra* spp).

Shallow sublittoral fine muds occur predominantly in extremely sheltered areas with very weak tidal currents. Such habitats are found in sea lochs and some rias and harbours. Populations of the lugworm *Arenicola marina* may be dense, with anemones, the opisthobranch *Philine aperta* and synaptid holothurians also characteristic in some areas. Characteristic species of circalittoral fine muds are sea pens, burrowing anemones and the ophiuroid *Amphiura* spp. The relatively stable conditions often lead to the establishment of communities of burrowing megafaunal species, such as *Nephrops norvegicus*.

3.7.2 Description of Fishing Activities

The commercial fishing activities likely to occur within these habitat groups are shown in Table 16.

Table 16 Commercial fishing activities associated with mud habitats

Habitat Group	Shore Access	Hand Gathering	Static Gears	Towed gears
H11	✓	✓	✓	✓
H11, Subtidal	✓	✓	✓	✓
H10, H11, H12	✓	✓	✓	✓
H10, H11, H13	✓	✓	✓	✓
H10, H11	✓	✓	✓	✓
H19			✓	✓

Intertidal muddy sediments may be subject to commercial harvesting of bivalves and other species. Cockles are the target species of fishers in intertidal and estuarine habitats and are harvested either mechanically (e.g. using suction dredges) or by large numbers of fishers using hand rakes (Kaiser et al. 2001).

Subtidally, fishing activities undertaken in muddy habitats using towed gears include otter trawling (for fish and invertebrates, such as *Nephrops norvegicus*), beam trawling and various dredging activities.

3.7.3 Reviews and Sensitivity Assessments

Many studies have investigated the impacts of fishing activities on the benthic communities of soft-sediments, including the mud habitats included in this section. Studies and reviews of the impacts of different fishing gears on these benthic habitats include: Hall et al (2008); Kaiser et al (2006); Kaiser et al (2002); Johnson (2002); and Thrush and Dayton (2002). Tyler-Walters and Arnold (2008) reviewed the impacts of trampling/access on intertidal benthic habitats.

The sensitivity of intertidal muds and sands, which support the gaper clam (*Mya arenaria*), intertidal muddy sands and intertidal muds to impacts caused by access to fishing grounds, was assessed by Tyler-Walters and Arnold (2008). Hall et al. (2008) assessed the sensitivity of subtidal muddy sands, sandy muds and muds using an adapted Beaumaris approach.

The Marine Life Information Network (MarLIN) has produced sensitivity assessments using the MarLIN approach (including factors relevant to fishing activities) for numerous littoral and sublittoral mud sediment habitat types and species found in mud sediments including (for example):

- *Echinocardium cordatum* and *Ensis* sp. in lower shore or shallow sublittoral muddy fine sand;
- *Arenicola marina* and synaptid holothurians in extremely shallow soft mud; and
- Sea pens and burrowing megafauna in circalittoral soft mud.

MarLIN assessments of sensitivity (intolerance and recoverability) are summarised in Appendix 1.

3.7.4 Resistance (Tolerance)

The direct impacts of different types of fishing gear are strongly habitat-specific. Kaiser et al. (2006) reported that soft-sediments, and especially muddy sands, were very vulnerable to fishing impacts, particularly to intertidal dredging which had the most severe initial impact. Otter trawling, beam trawling and scallop dredging also all produced a significant immediate impact on this habitat. The impact of these fishing activities differed between phyla; for example, in muddy sand intertidal raking had an initial impact on annelids and crustaceans but not molluscs, whereas otter trawling impacted crustaceans more severely than annelids and molluscs. In mud communities, otter trawling had a significant negative short-term impact, but in the long-term, a positive effect (an increase in mean abundance of benthic taxa) was detected.

Several other studies have indicated that intertidal mud habitat types are affected by certain fisheries activities. For example, trampling of intertidal muddy sand and mud habitats has been shown to adversely affect bivalves, reduce the abundance of some infauna and increase the abundance of opportunistic infaunal polychaetes and meiofauna (Tyler-Walters and Arnold, 2008). Commercial cockle harvesting using hand raking in these intertidal habitats have been shown to lead to a three-fold increase in the damage rate of undersized cockles compared to unfished sites (Kaiser et al., 2001), whilst the use of towed demersal gears in these intertidal areas is known to affect these habitats and their associated fauna (e.g. Cotter et al. 1997, Hall and Harding 1997, Ferns et al. 2000) including long-lived bivalve species, such as *Mya arenaria* (Hall et al., 2008).

The macrofauna and near-surface infauna of subtidal stable muddy sands, sandy muds and muds are susceptible to physical disturbance from bottom fishing gears (e.g. Kaiser et al. 1996; Ball et al. 2000, Bergman and van Santbrink 2000a; Hansson et al. 2000; Nilsson and Rosenberg 2003). Abundance, species diversity and richness all decrease as fishing intensity increases, even at the level of the meiofauna (e.g. Schratzberger and Jennings 2002) and alterations in the size structure of populations may occur e.g. heart urchin, *Echinocardium austral* (Thrush et al. 1998).

Thrush and Dayton (2002) reviewed studies that investigated the impacts of different types of towed gears on mud habitats (Thrush and Dayton, 2002) Reported effects included:

- Decreases in species richness and diversity (Sanchez et al. 2000; Ball et al., 2000);
- Changes in community structure, decreases in the density of common bivalves and polychaetes and increases in the density of nemerteans (Sparks-McConkey and Watling 2001); and
- Significant decreases in biomass and production in high pressure fishing areas on muddy sand (no significant effect in low pressure areas) (Jennings et al. 2001).

In contrast, Tuck et al. (1998) found that the number of species, individuals and diversity increased in fished areas.

Impacts of towed demersal gears in soft-sediment, including mud habitats, can include smothering of suspension feeding fauna through the resuspension of sediment by the fishing gears. Kaiser et al. (2006) found that otter trawling had the most severe affect on suspension feeders in muddy habitats, possibly reflecting the great depths to which otter doors penetrate the soft sediment habitat. Both suspension and deposit feeding fauna were negatively impacted by scallop dredging in muddy habitats.

There is limited information on the impacts of static gears, however the available literature suggests that the impact of pots and set nets, if deployed correctly, are of limited concern on subtidal stable muddy sands, sandy muds and muds (Hall et al 2008), although the sensitivity of erect epifauna to these activities is species dependant. Sea pens in Scottish sea lochs, which have been smothered or uprooted by *Nephrops* creels have been observed to re-establish themselves if in contact with muddy substrate (Eno et al. 2001). Of the three sea pen species, the tall sea pen, *Funiculina quadrangularis* is likely to be the most vulnerable to damage because of its brittle stalk and inability to retract into the sediment (ICES 2003).

Biogenic Features

The habitat complexity of these habitat types are increased through the burrows and mounds produced by the associated megafauna and towed demersal gears have been shown to remove such biogenic structures in subtidal muddy sand/mud habitats (Nilsson and Rosenberg, 2003).

Geomorphological/Sedimentary Features

Towed demersal gears have been shown to alter the sedimentary characteristics of subtidal muddy sand/mud habitats by penetration of the sediment (Ball et al. 2000). Beam trawls, scallop dredges and demersal trawls will potentially damage this habitat group to a greater degree than fishing activities utilising lighter towed gear (e.g. light demersal trawls and seines) (Hall et al. 2008).

Trawling alters the physical environment of the benthos by creating furrows or scar from trawl doors, scouring and flattening the seabed with ground rope and weights, and redistributing sediment and other material (Churchill 1989; Riemann and Hoffmann 1991; Schwinghamer et al. 1996).

3.7.5 Resilience (Recovery)

In a recent meta analysis of the impacts of different fishing activities on the benthic biota of different habitats, muddy sands were found to be very vulnerable to the impacts of fishing activities, with recovery times predicted to take years (Kaiser et al., 2006).

This long recovery time (from months to over a year for muddy sand, compared to days to months for sand) is due to the fact that these habitats are mediated by a combination of physical, chemical and biological processes (compared to sand habitats that are dominated by physical processes). Recolonization of the habitat is likely due, at least in part, to recolonization, requiring recruitment of larvae (as opposed to migration of adult organisms in sand habitats) (Kaiser et al., 2006).

Biogenic Features

Recovery of small-scale habitat complexity created by organisms will depend on the recovery rates of species populations. Short-lived species, such as amphipods, have rapid life cycles and hence recovery should be relatively rapid. However, for larger long-lived species recovery will take longer.

Geomorphological/Sedimentary Features

In a study comparing the responses of marine benthic communities within a variety of sediment types to physical disturbance, Dernie et al. (2003) found that muddy sand habitats had the longest recovery times, whilst mud habitats had an 'intermediate' recovery time (compared to clean sand communities which had the most rapid recovery rate).

Smith et al (2007) used side scan sonar and underwater video technology to record trawl impacts on silty clay sediment and found that trawl marks were evident throughout the year in the study area (including during the closed season). However, the trawl marks were less visible by the end of the close season (four months later) indicating that they had been biogenically weathered.

3.8 Saltmarsh

The habitat types (and corresponding CCW habitat groups and EUNIS classification) referred to in this review section are shown in Table 17.

Table 17 Saltmarsh habitat groups

Habitat types	CCW habitat group	EUNIS L3	EUNIS L4	EUNIS Name
Angiosperm-dominant soft littoral sediments, fisheries nursery grounds (Saltmarsh),	H13	A2.5	A2.51	Saltmarsh drift lines
	H13	A2.5	A2.52	Upper saltmarshes
	H13	A2.5	A2.53	Mid-upper saltmarshes and saline and brackish reed, rush and sedge beds
	H13	A2.5	A2.54	Low-mid saltmarshes
	H13	A2.5	A2.55	Pioneer saltmarshes

3.8.1 Brief Description of Habitat Group(s):

Saltmarshes comprise the upper vegetated portions of intertidal mudflats, lying approximately between mean high water neap and mean high water spring tide water levels. These habitats are usually restricted to relatively sheltered locations such as estuaries and saline lagoons. Saltmarsh vegetation is composed of a limited number of salt-tolerant species adapted to regular immersion, with more diverse plant communities found in the mid-upper marsh compared to the low-mid marsh.

Saltmarshes have high primary productivity, sustain diverse fish and mollusc assemblages, and are particularly important to fisheries as they function as nursery areas for juvenile fish.

3.8.2 Description of Fishing Activities

The commercial fishing activities likely to occur within these habitat groups are shown in Table 18.

Table 18 Commercial fishing activities associated with saltmarsh habitats.

Habitat Group	Shore Access	Hand Gathering	Static Gears	Towed gears
H13	✓	✓		

Saltmarsh occurs in the higher edges of the intertidal zone and therefore the only commercial fishing activities which may affect this habitat group are access to fishing grounds and hand gathering by professionals.

3.8.3 Reviews and Sensitivity Assessments

There is a relative paucity of information regarding the impacts of fishing activities on saltmarsh habitats. However, saltmarsh sensitivity to static gear, casual hand gathering and fishing grounds accessed by foot / vehicle was included in reviews of the impacts of access by Hall et al. (2008) and Tyler-Walters and Arnold (2008), and recreational and/or transportation ecology (Liddle 1991, Liddle 1997, Yorks et al. 1997, Davenport and Davenport 2006b, Davenport and Davenport 2006a, Davenport and Switalski 2006).

The Marine Life Information Network (MarLIN) has produced sensitivity assessments using the MarLIN approach (including factors relevant to fishing activities) for *Puccinella maritima* saltmarsh community and *Salicornia* sp. pioneer saltmarsh. MarLIN assessments of sensitivity (intolerance and recoverability) are summarised in Appendix 1.

3.8.4 Resistance

Localised damage to Atlantic salt meadow communities has been reported by CCW, for example, as a result of use of vehicles (primarily all-terrain vehicles) generally through people seeking access to harvest shellfish in the estuaries.

Saltmarsh is relatively resistant to foot trampling, although only a few passes of off-road vehicles can damage and remove the natural vegetation (Tyler-Walters and Arnold, 2008).

Saltmarsh was considered by Hall et al. (2008) to have a high sensitivity to professional harvesting as collection occurs over the entire saltmarsh and can be intensive.

Biogenic Features

Saltmarsh vegetation creates structurally complex habitat, damage and removal of vegetation will therefore reduce habitat complexity.

Geological/Sedimentary Features

Off-road vehicles can damage peat substratum underlying saltmarsh (Tyler-Walters and Arnold 2008).

3.8.5 Resilience (Recovery)

Studies in Californian saltmarshes have shown that recovery rates are species-specific and generally occur through vegetative in-growth of plants surrounding a disturbed spot or by growth of buried plants through the sediment. Seedling establishment was rare (Allison 1995).

Biogenic Features

Saltmarsh vegetation creates structurally complex habitats; recovery of this feature from the impacts described above will result in the restoration of a more structurally complex habitat.

Geological/Sedimentary Features

Off-road vehicles can damage peat substratum delaying recovery of saltmarsh (Tyler-Walters and Arnold, 2008).

3.9 Sand Sediments

The habitat types (and corresponding CCW habitat groups and EUNIS classification) referred to in this review section are shown in Table 19.

Table 19 Sand sediment habitat groups

Habitat types	CCW habitat group	EUNIS L3	EUNIS L4	EUNIS Name
Stable predominantly subtidal fine sands	H18	A2.2	A2.23	Polychaete/amphipod-dominated fine sand shores
	H18	A5.1	A5.12	Sublittoral coarse sediment in variable salinity (estuaries)
	H18	A5.2	A5.25	Circalittoral fine sand
Dynamic, shallow water fine sands	H24	A5.2	A5.22	Sublittoral sand in variable salinity (estuaries)
	H24	A5.2	A5.23	Infralittoral fine sand

3.9.1 Brief Description of Habitat Group(s):

Sandy habitat types are typically characterised by animals living within the sediment (infauna), rather than attached epifauna and epiflora, although some species may have structures that protrude above the surface, e.g. polychaete and amphipod tubes, adding to the complexity of the habitat. *Sabellaria* reefs are more structurally complex habitats associated with sandy substrates and these are considered in the review of biogenic reefs (Section 3.1).

The type of biological assemblage that develops at a location is primarily influenced by sediment characteristics, which in turn depend on the prevailing hydrodynamic conditions.

Coarse sand sediment occurring in sand-wave formations in shallow water, wave exposed and tide-swept coasts are mobile sediments subjected to high levels of natural disturbance. The infauna in this type of habitat is highly impoverished and is typified by small opportunistic capitellid and spionid polychaetes and isopods that are adapted to living in such a highly perturbed environment. The epifauna is characterised by mobile predators such as crabs, hermit crabs (*Pagurus bernhardus*), whelks and occasionally sand eels (*Ammodytes* spp.).

Loose, coarse sand habitats fully exposed to wave action and swept by strong tidal streams are dominated by small or highly mobile polychaetes, thick shelled and rapidly burrowing bivalves (*Spisula elliptica* and *S. subtruncata*) and mobile amphipods that are adapted to periodic disturbance.

Shallow areas with coarse sand swept by tidal currents but sheltered from wave exposure may develop dense beds of the sand mason polychaete *Lanice conchilega.* The biogenic structures created by these organisms increase habitat complexity and influence physical parameters, for example reducing near-bed currents and significantly increasing sediment stability. Larsonneur (1994) reported that sand stabilised by sand masons is sufficiently stable to allow subsequent colonization by *S. alveolata.*

A close variant of this community occurs in fine compacted sands with moderate exposure and weak tidal currents. This habitat is characterised by the thin-shelled bivalve *Fabulina fabula*, and is found in the Irish Sea, north-east coast of England and in numerous Scottish sea lochs (JNCC 2009).

3.9.2 Description of Fishing Activities

The commercial fishing activities likely to occur within these habitat groups are shown in Table 20.

Table 20 Commercial fishing activities associated with sand sediment habitats

Habitat Group	Shore Access	Hand Gathering	Static Gears	Towed gears
H18			✓	✓
H24			✓	✓

Most flatfish fisheries are found in areas of sandy seabed and are therefore subjected to intensive perturbation by bottom fishing gears (such as beam trawling) in the southern North Sea and English Channel (JNCC 2009).

Some of the bivalve species found in these habitats, such as *Pecten maximus,* are subject to significant fishing effort. Other species, such as *Paphia rhomboides, Glycymeris glycymeris, Chamelea gallina,* and *Ensis* spp are only subject to occasional fishing effort. Most of these species are exported to continental Europe for human consumption (JNCC 2009).

Static gears, such as crab pots, may be used in these habitats, although probably to a lesser degree compared to rocky habitats.

3.9.3 Reviews and Sensitivity Assessments

Many studies have investigated the impacts of fishing activities on the benthic communities of soft-sediments, and the majority of these studies have occurred on sand habitats (Kaiser et al., 2006), including the fine sand habitats included in this review section, but also coarse sand (reviewed Section 3.6 – mixed sediments) and muddy sands (reviewed in Section 3.7 - mud). A detailed review of all of these studies is beyond the scope of this report. However, studies and reviews of the impacts of different fishing gears on these benthic habitats include: Hall et al (2008), Kaiser et al (2006), Kaiser et al. (2002), Johnson (2002), Thrush and Dayton (2002) and Collie et al. (2000).

The relative sensitivity of different seabed types and associated benthic species to fishing disturbance was assessed by Macdonald et al. (1996). More recently, the sensitivity of these sand habitat groups to commercial fishing impacts was assessed by Hall et al. (2008) using the adapted Beaumaris approach.

The Marine Life Information Network (MarLIN) has produced sensitivity assessments using the MarLIN approach (including factors relevant to fishing activities) for sand sediments including:

- *Amphiura filiformis* and *Echinocardium cordatum* in circalittoral clean or slightly muddy sand;
- *Virgularia mirabilis* and *Ophiura* spp. on circalittoral sandy or shelly mud;
- *Neomysis integer* and *Gammarus* spp. in low salinity infralittoral mobile sand.

MarLIN assessments of sensitivity (intolerance and recoverability) are summarised in Appendix 1.

3.9.4 Resistance (Tolerance)

The impacts of fishing activities on benthic communities varies with gear type, habitat and between taxa (Collie et al. 2000; Thrush and Dayton, 2002 and Kaiser et al. 2006). Studies investigating the biological impacts of various towed gears on sand habitats were reviewed by Thrush and Dayton (2002). Gear type and habitat type influenced the severity of the effect on benthic communities with several of the studies indicating that certain fishing activities had no detectable impacts on specific habitat types, including Kaiser and Spencer (1996; beam trawling in unstable sand habitats), Kenchington et al. (2001; otter trawling on sand) and Van Dolah et al. (1991; shrimp trawling on sand). Similarly, Kaiser et al. (2006), who undertook a meta-analysis to examine the response of benthic biota in different habitats to different fishing gears, showed that the direct impacts of different types of fishing gear are strongly habitat-specific as some habitats are pre-adapted to natural disturbance and are characterised by species that are relatively resistant or can recover rapidly.

The epifauna and infaunal assemblages of both stable and dynamic fine sands are susceptible to direct physical disturbance from towed demersal gears and dredges which penetrate and disturb the sediment e.g. Eleftheriou and Robertson 1992; Kaiser et al. 1998; Robinson and Richardson 1998; Schwinghamer et al. 1996; Freese et al. 1999; Prena et al. 1999; Bergman and Van Santbrick 2000a,b; Tuck et al. 2000; Kenchington et al. 2001; Gilkinson et al. 2005, all cited in Hall et al 2008. In general, fishing using towed gears results in the mortality of non-target organisms either through physical damage inflicted by the passage of the trawl or indirectly by disturbance, damage, exposure and subsequent predation. Beam trawling, for example, decreases the density of common echinoderms, polychaetes and molluscs (Bergman and Hup, 1992) and decreases the density and diversity of epifauna in stable sand habitats (Kaiser and Spencer, 1996).

Other reported effects of towed gears on benthic communities include:

- short-term decreases in biomass and abundance of macrofauna (Pranovi et al. 2000; Veale et al. 2000, Watling et al. 2001).

- lower diversity (McConnaughey et al. 2000, Veale et al., 2000);
- altered species composition (Engel and Kvitek, 1998)

The impact of fishing activities also differs between phyla; beam trawling in sand habitats has been shown to have a greater initial impact on crustaceans, echinoderms and molluscs compared to annelids (Kaiser et al., 2006). These differences are due to position in the habitat and ability to withstand physical damage. Bergman and van Santbrink (2000a,b) showed that the single passage of a beam trawl (4 m or 12 m) or an otter trawl in sandy or silty areas caused direct mortality of about 5-40 per cent for some gastropod, starfish, crustacean and annelid worm species and between 20-65 per cent for some bivalve species. Studies show that sedentary or attached macrofauna are more vulnerable to fishing impacts (McConnaughey et al., 2000, Engel and Kvitek, 1998, Eleftheriou and Robertson, 1992). The populations of larger, longer-lived species are less resistant to fishing impacts than smaller, short-lived species that are able to compensate for increased mortality.

Compared to towed gear, benthic communities are relatively unaffected by static fishing gears (fish or crustacean pots, long-lines or anchored nets) due to the relatively small area of seabed directly affected (Kinnear et al. 1996; Jennings and Kaiser 1998; Eno et al. 2001). Benthic community biomass in areas subjected to only static gear use has been reported to be significantly greater compared to areas in which trawling has occurred within the last two years (Blyth et al. 2004). It is possible that any epifauna present may be damaged and/or detached on contact with static gears and the ability of epifauna to resist impacts from static gears varies between species (see Section 3.10). The degree of impact will depend on the intensity of the fishing and the duration. Weights and ropes associated with static gears also have the potential to damage, entangle or remove epifaunal species and where the gear drags or bounces the damage will be more widespread.

Biogenic Features

In soft sediment habitats, much of the structural habitat complexity is produced by animals. Towed fishing gears can damage/kill and/or remove such organisms and flatten their biogenic structures and hence remove this habitat complexity (Thrush and Dayton 2002).Towed gears can also remove small scale physical habitat complexity such as sand ripples (e.g. Auster et al. 1996).

Geomorphological/Sedimentary Features

Intertidal dredging and raking have some of the most severe impacts on soft-sediments including sand habitats (Kaiser et al., 2006). Intertidal dredging leads to the physical removal of and resuspension of substratum in the water column (furrows of up to 10 cm deep; Kaiser et al. 2006).

Towed demersal gear alters the sedimentary habitats of fine sands by penetrating the sediment, smoothing the habitat (Schwinghamer et al. 1996, 1998, cited in Hall et al 2008) and smothering habitat features by re-suspending sediments in the water column (Jennings and Kaiser 1998). Lighter towed gear e.g. light demersal trawls and seines, have less impact (Drabsch et al. 2001).

3.9.5 Resilience (Recovery)

Fine sands are characterised by robust fauna which could potentially recolonise habitats after disturbance events (Hall et al. 2008). For sand habitats that are dominated by physical processes, habitat restoration (post-fishing activity) is relatively rapid (days to a few months) and recolonization is probably dominated by active and passive migration of adult organisms into the disturbed areas (e.g. McLusky et al., 1983 cited in Kaiser et al. 2006). However, some sandy sediment communities also contain large bodied, slow growing fauna, such as the bivalves *Mya truncate* and *Arctica islandica*, which are sensitive to fishing disturbances and are likely to have long recovery periods (e.g. Witbaard and Bergman 2003 and Beukema 1995).

In a study comparing the responses of marine benthic communities within a variety of sediment types to physical disturbance, Dernie et al. (2003) found that clean sand communities had the most rapid recovery rate following disturbance.

Biogenic Features

Recovery of small-scale biogenic features will depend on the recovery of species populations. Recovery of short-lived species such as amphipods which have rapid life cycles should be relatively rapid but will take longer for larger, long-lived species.

Geomorphological/Sedimentary Features

In areas of strong water movement, the recovery of soft sediment and sediment features is dependent on the prevailing hydrodynamic conditions but may be expected to be rapid where sediments are mobile.

Schwinghamer et al. (1996) examined the effect of otter trawls on habitat with sand substrate (fine and medium grained sand) in the Grand Banks one and two years after trawling had stopped. The tracks left by the trawl doors were visible for at least ten weeks but not visible or only faintly visible after one year.

3.10 Slow Growing Epifauna

The habitat types (and corresponding CCW habitat groups and EUNIS classification) referred to in this review section are shown in Table 21.

3.10.1 Brief Description of Habitat Group(s)

These habitat groups are characterised by slow-growing epifauna on subtidal rock. These assemblages include a variety of fauna including erect and branching species, which are characteristically slow growing and vulnerable to physical disturbance due to their growth form. These communities also tend to be species rich. Typical species include axinellid sponges, pink sea fan (*Eunicella verrucosa*) and ross (*Pentapora foliacea*). Soft rock biotopes are also included in this habitat.

Epiflora such as maerl and macroalgae and are considered in separate reviews (see Sections 3.1 and 3.5 respectively). Fast growing epifauna such as hydroids and bryozoans are also considered separately (see Section 3.4).

Table 21 Slow growing epifauna dominated habitat groups

Habitat types	CCW habitat group	EUNIS L3	EUNIS L4	EUNIS Name
Slow growing erect and branching subtidal species	H15	A4.1	A4.13	Mixed faunal turf communities on circalittoral rock
Slow growth predominantly subtidal rock with erect and branching species	H21	A3.3	A3.35	Faunal communities on low energy infralittoral rock
	H21	A4.1	A4.11	Very tide-swept faunal communities on circalittoral rock
	H21	A4.2	A4.21	Echinoderms and crustose communities on circalittoral rock
	H21	A4.2	A4.23	Communities on soft circalittoral rock

3.10.2 Description of Fishing Activities

The commercial fishing activities likely to occur within these habitat groups are shown in Table 22.

Table 22 Commercial fishing activities associated with slow growing epifauna dominated habitats

Habitat Group	Shore Access	Hand Gathering	Static Gears	Towed gears
H15			✓	✓
H21			✓	✓

This habitat group is located in the subtidal region so only the impacts of static and towed gears are relevant to this review. In general, towed gear is generally not a major threat to rocky habitat types, as they are unsuitable for both trawls and dredges. However there are types of towed gear designed for rocky areas - the rock hopper otter trawl, and the Newhaven scallop dredge - and these could pose a risk to this habitat group where it occurs on gently sloping or level rock.

Static gear is deployed regularly on rocky grounds, either in the form of pots or creels, or as bottom set gill or trammel nets. Whilst the potential for damage is lower per unit deployment compared to towed gear, there is a risk of cumulative damage to sensitive species if use is intensive. Damage could be caused during the setting of pots or nets and their associated ground lines and anchors and by their movement over the bottom during rough weather and during recovery.

3.10.3 Reviews and Sensitivity Assessments

MacDonald et al (1996) reviewed the sensitivity of seabed types and benthic species, including several species of slow growing epifauna, to a ‘single encounter’ with static and towed fishing gears. The sensitivity of these habitats to fishing activities was also more recently reviewed by Hall et al. (2008) for CCW.

MarLIN has produced sensitivity assessments using the MarLIN approach (including factors relevant to fishing activities) for various slow growing epifaunal species including *Eunicella verrucosa*, *Leptopsammia pruvoti*, *Pentapora foliacea*, *Echinus esculentus* and *Alcyonium digitatum*. They have also assessed the sensitivity of a number of habitat types in which slow growing epifauna are found:

- Erect sponges, *Eunicella verrucosa* and *Pentapora foliacea* on slightly tide-swept moderately exposed circalittoral rock
- Faunal and algal crusts, *Echinus esculentus*, spare *Alcyonium digitatum* and grazing-tolerant fauna on moderately exposed circalittoral rock.

MarLIN assessments of sensitivity (intolerance and recoverability) are summarised in Appendix 1.

3.10.4 Resistance (Tolerance)

Emergent organisms can be tangled, damaged or removed by the passage of towed gears or the setting and hauling of static gears and anchors. Fragile or brittle organisms have relatively low resistance to a physical strike by fishing gears.

Fishing using towed gears reduces the density of long-lived epifauna (Thrush et al. 1998). Scallop dredging has been shown to reduce the abundance of sponges leading to long term changes in the structure and biodiversity of sponge assemblages (Kefalas et al. 2003). Some organisms may pass safely under towed gears although potentially individuals can be torn from the substrate or broken up. Size may partially determine resistance to the impact as larger species are less likely to pass unscathed under gears (Wassenberg et al. 2002). Resistance to trawling also varies between species (Wassenberg et al. 2002).

MacDonald et al. (1996) assessed the sensitivity of different benthic species to fishing disturbance by ‘scoring’ each species fragility (its ability to withstand the physical impact of a single fishing disturbance) and recovery potential (assuming no further fishing disturbance occurred). The slow growing epifaunal species *Leptopsammia pruvoti*, *Eunicella verrucosa*, *Pentapora foliacea* and *Echinus esculentus* were classified as being ‘very fragile’, whilst *Caryophyllia smithii* was classified as ‘moderately fragile’ and *Alcyonium digitatum* and *Pomatoceros triqueter* were classified as ‘not very fragile’.

Some epifauna may be relatively resistant to potting activities. *Eunicella* have been shown to flex as creel pots are hauled over them and to spring back when released (Eno et al. 2001). Others may be less resistant and *Pentapora* have been shown to be badly damaged by direct hits from pots (Eno et al. 2001). Epifauna will also be damaged where they are rubbed by ropes (Hall et al. 2008). Observations in Lyme bay have shown that pots can be dragged by wind and tidal currents and, where the amount of line is insufficient strong swells can cause the weights to bounce on the seabed causing damage (Eno et al. 2001).

Biogenic Features

Much of the structural complexity of this habitat type is provided by the emergent epifauna, such that removal of, or damage to, these organisms arising from commercial fishing activities will reduce the degree of biogenic structure.

Geomorphological/Sedimentary Features

Hard substrates are relatively resistant to physical damage from fishing gears. However, towed gears may damage softer rock types or may physically damage the substrate, reducing complexity.

3.10.5 Resilience (Recovery)

Recovery will depend on the life-history characteristics of the species affected, including the ability of damaged adults to repair/regenerate lost or damaged parts and/or the ability of larvae to reach and recolonise the habitat. Re-establishment of long-lived, slow-growing species in which maturity occurs late will be slower than for smaller species with faster life cycles. Colonial organisms such as sponges may have good regenerative abilities, able to regenerate tissue rapidly from small fragments (Fish and Fish, 1989 cited in McMath et al. 2000). Populations of some sessile species may rely on spawning events to allow recolonization. Where fishing frequently occurs, the time between fishing events may not be great enough to allow re-establishment of the assemblage.

An example of how recovery may vary between species is highlighted by Macdonald et al. (1996) who assessed the sensitivity of different benthic species to fishing disturbance by ‘scoring’ each species fragility (ability to withstand physical impact of a single fishing disturbance) and recovery potential (assuming no further fishing disturbance occurred). Several slow growing epifaunal species were assessed including: *Leptopsammia pruvoti* (scored as having a ‘very long recovery/no recovery likely), *Eunicella verrucosa* and *Caryophyllia smithii* (scored as having a ‘long’ recovery potential), *Pentapora foliacea Alcyonium digitatum* and *Echinus esculentus* (scored as having ‘moderate’ recovery potential) and Pomatoceros triqueter (scored as ‘short’ recovery potential).

Biogenic Features

Much of the structural complexity of this habitat type is provided by the emergent epifauna, such that the recovery of these organisms arising from commercial fishing activities will increase the degree of biogenic structure.

Geomorphological /Sedimentary Features

Hard substrates are predicted to have low, if any, recoverability from physical damage.

3.11 Chalk Reefs

The habitat types (and corresponding CCW habitat groups and EUNIS classification) referred to in this review section are shown in Table 23.

3.11.1 Brief Description of Habitat Group(s):

This habitat group includes caves and overhangs in limestone within the littoral zone. However, caves and overhangs are protected (by the fact of their location) from static and towed gears and hence are not considered further in this section.

Table 23 Chalk reef habitat groups

Habitat types	CCW habitat group	EUNIS L3	EUNIS L4	EUNIS Name
Chalk reefs	none	A1.4	A1.44	Communities of littoral caves and overhangs
	none	A3.2	A3.21	Kelp and red seaweeds (moderate energy infralittoral rock)
	none	A4.2	A4.23	Communities on soft circalittoral rock

Factors influencing the community assemblages within this habitat group include the amount of scour, wave surge, and the degree of light penetration. Sublittoral soft chalk is often too soft for sessile filter-feeding animals to attach and thrive in large numbers, hence an extremely impoverished epifauna results on upward-facing surfaces, although vertical faces may be somewhat richer. The rock is sufficiently soft to be bored by bivalves. Benthic communities on soft, moderately wave exposed circalittoral bedrock with moderately strong tidal streams are dominated by the piddock *Pholas dactylus*. Other species present typically include the polychaete *Polydora* and *Bispira volutacornis*, the sponges *Cliona celata* and *Suberites ficus*, the bryozoan *Flustra foliacea*, *Alcyonium digitatum*, the starfish *Asterias rubens*, the mussel *Mytilus edulis* and the crab *Necora puber* and *Cancer pagurus*. Foliose red algae may also be present.

Chalk reefs sustain a rich community of reefs encrusted with kelp, red algae, ‘boring’ sponges, baked bean sea squirts and dead men’s fingers. They are rich feeding grounds for fish and cuttlefish and provide nursery grounds for shark such as tope or smooth-hound, and black sea bream.

3.11.2 Description of Fishing Activities

The commercial fishing activities likely to occur within these habitat groups are shown in Table 24.

Table 24 Commercial fishing activities associated with chalk reefs

Habitat Group	Shore Access	Hand Gathering	Static Gears	Towed gears
Chalk Reefs			✓	✓

Static gear is deployed regularly on rocky grounds, either in the form of pots or creels, or as bottom set gill or trammel nets. Whilst the potential for damage is lower per unit deployment compared to towed gear, there is a risk of cumulative damage to sensitive species if use is intensive. Damage could be caused during the setting of pots or nets and their associated ground lines and anchors, and by their movement over the bottom during rough weather and during recovery.

In general, fishermen using towed gears will avoid areas where there is a risk of snagging (as this can result in the loss of the gear and place the vessel and crew at risk). This is likely to reduce disturbance from this source in this habitat type.

3.11.3 Reviews and Sensitivity Assessments

No literature specifically addressing the impacts of fishing activities on this group of habitats was sourced as part of this review. As such, impacts have been inferred from other habitat groups, particularly faunal turfs (Section 3.4) and slow growing epifauna (Section 3.10).

The Marine Life Information Network (MarLIN) has produced sensitivity assessments using the MarLIN approach (including factors relevant to fishing activities) for characteristic species found within these habitat groups, including *Pholas dactylus*, *Mytilus edulis*, *Alcyonium digitatum* and *Flustra foliacea*. They have also assessed the sensitivity for the following chalk reef habitat types:

- *Rhodothamniella floridula* in littoral fringe soft rock caves;
- Piddocks with a sparse associated fauna in upward-facing circalittoral very soft chalk or clay;
- *Laminaria digitata* and piddocks on sublittoral fringe soft rock; and
- *Polydora* sp. tubes on upward-facing circalittoral soft rock.

MarLIN assessments of sensitivity (intolerance and recoverability) are summarised in Appendix 1.

3.11.4 Resistance (Tolerance)

Some infauna may be relatively resistant to fishing impacts as their environmental position confers protection. Species that are able to bore into chalk reefs for example, piddocks and the boring sponge *Cliona celata*, are predicted to be relatively unaffected by fishing using static gears or towed gears that do not damage the reef.

In general, encrusting, sessile epifauna are known to be vulnerable to removal and damage by towed gears (McConnaughey et al., 2000; Engel and Kvitek, 1998; Eleftheriou and Robertson, 1992). In addition, the soft rock of chalk reefs may be abraded by towed gears, and epifauna may be damaged and removed.

Emergent organisms could be tangled, damaged or removed by the setting and/or hauling of static gears (pots, long-lines or anchored nets) and anchors. However, in general, compared to towed gear, benthic communities are relatively unaffected by static gears (pots, long-lines or anchored nets) due to the relatively small area of seabed directly affected (Kinnear et al. 1996; Jennings and Kaiser 1998; Eno et al. 2001). The ability of epifauna to resist impacts from static gears varies between species (see Section 3.10) and the degree of impact will depend on the intensity of the fishing and the duration.

MacDonald et al. (1996) assessed the sensitivity of different benthic species to fishing disturbance by ‘scoring’ each species fragility (its ability to withstand the physical impact of a single fishing disturbance) and recovery potential (assuming no further fishing disturbance occurred). Species assessed which occur in this habitat type included: *Cliona celata* and *Flustra foliacea* (both classed as ‘moderately fragile’) and *Alcyonium digitatum* (classified as ‘not very fragile’).

Biogenic Features

Much of the structural complexity of this habitat type is provided by the emergent epifauna and flora (e.g. tunicates, algae, mussels, bryozoans), and removal or damage to these organisms from commercial fishing activities will reduce the degree of biogenic structure and hence the complexity of the habitat for other animals.

Geomorphological/Sedimentary Features

Soft rock may be vulnerable to abrasion and erosion and hence towed gears may physically damage the substrate, reducing complexity, with low levels, if any, of recoverability from this impact.

Resilience (Recovery)

Recovery will be dependent on the ability of species to regenerate or recolonise. Re-establishment of long-lived, slow-growing species will be slower compared to smaller species with faster life cycles. Colonial organisms such as sponges have good regenerative abilities, able to regenerate tissue rapidly from small fragments (Fish and Fish, 1989, cited in McMath et al. 2000). Populations of some sessile species may rely on spawning events to allow recolonization. Where fishing frequently occurs, the time between fishing events may not be great enough to allow re-establishment of the assemblage.

MacDonald et al. (1996) assessed the sensitivity of different benthic species to fishing disturbance by ‘scoring’ each species fragility and recovery potential (assuming no further fishing disturbance occurred). Species assessed which occur in this habitat type included: *Cliona celata*, *Flustra foliacea* and *Alcyonium digitatum* which were also scored as having a ‘moderate’ recovery potential.

Biogenic Features

Much of the structural complexity of this habitat type is provided by the epifauna and flora, so that recovery of the degree of biogenic structure/complexity is dependent on recovery of these components.

Geomorphological/Sedimentary Features

Hard substrates are predicted to have low, if any, recoverability from physical damage.

3.12 Vertical and underboulder surfaces

The habitat types (and corresponding CCW habitat groups and EUNIS classification) referred to in this review section are shown in Table 25.

3.12.1 Brief Description of Habitat Group(s):

Vertical hard substrates typically host an assemblage of attached epifauna. The orientation and depth will determine the level of light penetration and hence macroalgal colonization. The substrate may also support some burrowing animals such as piddocks. Depending on the species present in these habitat types, the sensitivity reviews for faunal turfs (Section 3.4), macroalgae (Section 3.5), chalk reefs (Section 3.11) and slow-growing epifauna (Section 3.10) may be relevant to the current section.

Table 25 Vertical and underboulder habitat groups

Habitat types	CCW habitat group	EUNIS L3	EUNIS L4	EUNIS Name
Underboulder communities on lower shore and shallow subtidal boulders and cobbles	H26	A3.2	A3.21	Kelp and red seaweeds (moderate energy infralittoral rock)
	H26	A3.2	A3.22	Kelp and seaweed communities in tide-swept sheltered conditions
Vertical subtidal rock with associated community	H14	A3.7	A3.74	Caves and overhangs in infralittoral rock
	H14	A4.1	A4.13	Mixed faunal turf communities on circalittoral rock
	H14	A4.2	A4.21	Echinoderms and crustose communities on circalittoral rock

3.12.2 Description of Fishing Activities

Static gear is deployed regularly on rocky grounds, either in the form of pots or creels, or as bottom set gill or trammel nets (Table 26). Whilst the potential for damage is lower per unit deployment compared to towed gear, there is a risk of cumulative damage to sensitive species if use is intensive. Damage could be caused during the setting of pots or nets and their associated ground lines and anchors, and by their movement over the bottom during rough weather and during recovery.

Generally, steep and rocky substrata are unsuitable for both trawls and dredges. Fishermen using towed gears will generally avoid areas where there is a risk of snagging (as this can result in the loss of the gear and places the vessel and crew at risk). This is likely to reduce disturbance from this source in these habitat types. However there are types of towed gear designed for rocky areas and these could pose a risk to communities on vertical and underboulder surfaces, where these occur in fished areas. This habitat group also includes caves and overhangs in infralittoral rock that are generally unsuitable areas for fishing and hence are unlikely to be directly impacted by any fishing activity (e.g. Sewell and Hiscock, 2005).

Table 26 Commercial fishing activities associated with vertical and underboulder habitats

Habitat Group	Shore Access	Hand Gathering	Static Gears	Towed gears
H26			✓	✓
H14			✓	

3.12.3 Reviews and Sensitivity Assessments

The effects of fishing on habitats occurring within European Marine Sites, including submerged and partially submerged sea caves was assessed by Sewell and Hiscock (2005). The sensitivity of this underboulder habitat to fishing activities was assessed by Hall et al. (2008) for CCW.

The Marine Life Information Network (MarLIN) has produced sensitivity assessments using the MarLIN approach (including factors relevant to fishing activities) for the following habitat types:

- Underboulder communities;
- Sponge crusts and anemones on wave-surged vertical infralittoral rock; and
- *Alcyonium digitatum* and a bryozoan, hydroid and ascidian turf on moderately exposed vertical infralittoral rock.

MarLIN assessments of sensitivity (intolerance and recoverability) are summarised in Appendix 1.

3.12.4 Resistance (Tolerance)

Hall et al. (2008) indicated that subtidal underboulder communities were sensitive to low levels of exposure (a single pass) of towed gear. Disturbance could move boulders, crushing associated animals, or leaving them exposed to predators and environmental stressors such as desiccation. Intertidal boulder habitats were also considered sensitive to hand gathering and trampling activities which disturb boulders and rocks (Hall et al. 2008).

Static gears, such as pots, may impact this habitat during setting or hauling. For example, Hall et al. (2008) reported that potting activities in Lundy were banned from vertical rock surfaces due to the damage caused during hauling activities.

Biogenic Features

Some of the structural complexity of this habitat type is provided by the emergent epifauna or macroalgae, so that removal or damage to these organisms from commercial fishing activities will reduce the degree of biogenic structure present.

Geomorphological/Sedimentary Features

Hard substrates are relatively resistant to physical damage from fishing gears. However, towed gears may damage softer rock types or may physically damage the substrate, reducing complexity.

Fishing activities may scatter boulders reducing the physical complexity of the habitat (piled boulders create structurally complex habitats with a range of micro-habitat features which provide shelter in gaps and crevices).

3.12.5 Resilience (Recovery)

The recovery rate of the biological assemblage depends on the life history characteristics of the species present including their ability to repair damage and recover from impacts. If species are removed and killed then the rate of recovery may depend on nearby sources of colonists. The most relevant data comes from the studies of Sebens (1985, 1986) in the USA from artificial reef colonization experiments and life history studies.

The recolonization of epifauna on vertical rock walls was investigated by Sebens (1985, 1986). He reported that rapid colonizers such as encrusting corallines, encrusting bryozoans, amphipods and tubeworms recolonized within 1-4 months. Ascidians such as *Dendrodoa carnea*, *Molgula manhattensis* and *Aplidium* spp. achieved significant cover in less than a year, and, together with *Halichondria panicea*, reached pre-clearance levels of cover after two years. A few individuals of *Alcyonium digitatum* and *Metridium senile* colonized within four years (Sebens 1986) and would probably take longer to reach pre-clearance levels.

Jensen et al. (1994) reported the colonization of an artificial reef in Poole Bay, England. They noted that erect bryozoans, including *Bugula plumosa*, began to appear within six months, reaching a peak in the following summer, 12 months after the reef was constructed. Similarly, ascidians colonized within a few months e.g. *Aplidium* spp. Sponges were slow to establish, with only a few species present within 6-12 months but beginning to increase in number after two years, while anemones were very slow to colonize with only isolated specimens present after two years (Jensen et al. 1994.). In addition, Hatcher (1998) reported a diverse mobile epifauna after one year's deployment of the settlement panels.

Overall, bryozoans, hydroids, and ascidians are opportunistic, grow and colonize space rapidly and will probably develop a faunal turf within 1-2 years. Mobile epifauna and infauna will probably colonize rapidly from the surrounding area. However, slow growing species such as some sponges and anemones, will probably take many years to develop significant cover, so that a diverse community may take up to 5 -10 years to develop, depending on local conditions (Tyler-Walters 2008).

Biogenic Features

Much of the biogenic structural complexity of this habitat type is provided by epifauna and flora, so that recovery of the degree of biogenic structure/complexity is dependent on recovery of these components.

Geomorphological/Sedimentary Features

Hard substrates are predicted to have low recoverability from physical damage. If piled boulders are scattered they will remain this way unless physical conditions such as wave action are sufficient to move them (this may depend on storms or other infrequent events) or they are actively restored through human intervention.

4 Habitat parameters and ecological attributes (Task 2)

The earliest approaches to marine habitat sensitivity assessment relied primarily on the physical characteristics of the shoreline (Gundlach and Hayes 1978; Weslawski et al. 1997), as these were the primary characteristics affecting the physical effects of oil spills on the shore. Weslawki et al. (1997) also included biological characteristics, focusing on potentially sensitive communities, limited to macrophyte cover, amphipod density, bird moulting areas, seabird feeding grounds and seal haul-outs, together with an estimate of recovery or resettlement potential.

Seminal work by MacDonald et al. (1996), Holt et al. (1995, 1997) and Hiscock (1999) demonstrated the need to consider physical and biological characteristics of the habitat or species. This work was built on by SENSMAP[2] and MarLIN (Hiscock et al. 1999;Tyler-Walters and Jackson 1999; McMath et al. 2000; Tyler-Walters et al. 2001) who incorporated physical, chemical and biological traits of both species and habitat to assess their sensitivity to 24 separate pressures likely to be caused by natural events and human activities.

Recent studies of the effect of fishing have strived to develop empirical estimates of the relative sensitivity of marine habitats to the effects of fishing activities (Kaiser et al. 2006; Hiddink et al. 2007). In doing so, they have focused on a number of relevant biological traits, and tried to evaluate those that detect impact and/or relative sensitivity. In the same period, the rise of biological traits analysis has applied numerous traits (using multivariate techniques) to the study of marine communities, in order to determine the effects of fishing activities and other pressures on community structure and function.

4.1 Parameters / ecological attributes used

This review looked at the wide variety of parameters used in a number of published studies and reviews. A total of 130 parameters were derived from 70 studies. The parameters used are summarised in Table 27 and listed in full in Appendix 2.

In the studies reviewed, parameters were characterised into:

- physical (morphological)
- chemical, and
- biological descriptors.

These parameter groups can be further subdivided into descriptors of:

- the fishing activity;
- the biological community,
- the species;
- reproductive and life history descriptors; and
- specific habitats.

[2] Sensitivity mapping of inshore marine biotopes in the southern Irish Sea (SensMap)

Therefore, Table 27 is subdivided (a-d) accordingly, and cross referenced to the relevant habitat types examined in each case.

Parameters that were specific to habitat type included percentage maerl cover and mean maximum size of rhodolith (maerl), leaf width and number of leaves (seagrass), and phenology and photosynthetic pathway (saltmarsh).

Table 27a Fishing activity parameters / attributes used in prior studies of sensitivity

Descriptors of fishing activity	Biogenic	Sand	Mud	Mixed	Seagrass	Faunal Turfs	Macroalgae	Saltmarsh	Slow growing epifauna
Area of impact					x				
Catch rate				x					
Catch weight				x					
Damage rate of cockles			x						
Efficiency of capture (%)				x					
Fishing effort		x	x	x	x				
Frequency of shell disturbance marks		x							
Gear type	x	x	x						x
Individuals per tube head				x					
Mean catch rate						x			
Mean Damage Index (MDI)				x					
Mesh size				x					
Mortality (no killed/population)	x	x	x	x					
Number of charismatic animals caught[1]				x					
% Taxa damaged						x			
% Total area dredged			x						
Trampling intensity from shore access to fishing grounds (no./duration/weight)	x					x		x	
Trawl door tracks			x						

Notes:
[1] Refers to parameter included in model system (see Fulton *et al*. 2005)
[2] Refers to parameters used in the MarLIN sensitivity approach (www.marlin.ac.uk/habitats.php).

Table 27b Physical (morphological) parameters / attributes used in prior studies of sensitivity

Physical (morphological)	Biogenic	Sand	Mud	Mixed	Seagrass	Faunal Turfs	Macroalgae	Saltmarsh	Slow growing epifauna
Area covered by maerl thalli	x								
Bottom stress[1]				x					
Burrowing rate		x							
% Canopy cover					x				
% Covering of dead matte (maerl)					x				
Density and size of mounds			x						
Depth			x	x	x				
Depth distribution of infauna			x						
Depth of furrows		x							
Depth of trawl		x							
Depth of water within disturbed pits		x	x	x					
Diet analysis		x		x					
Dissolved oxygen	x								
Distribution of patches					x				
Disturbance level				x					
Erosion rate[1]				x					
Grain-size analysis		x	x	x	x				
Habitat recovery rate (mm/day)		x	x	x					
Latitude								x	
Light[1]				x					
Load bearing capacity	x								
% Maerl cover	x								
Mean dry weight of sediment	x								
Mean maximum size of rhodolith	x								
No. and volume of tube heads per sample				x					
Rate of sediment erosion	x								
Reef height	x								
Physical changes to seafloor			x						
Seabed topography			x						

Physical (morphological)	**Biogenic**	**Sand**	**Mud**	**Mixed**	**Seagrass**	**Faunal Turfs**	**Macroalgae**	**Saltmarsh**	**Slow growing epifauna**
Seabed type[1]				x					
Season				x					
Sediment composition (organic content, % silt and clay content, water content)	x	x	x	x					
Sediment flux				x					
Sediment infilling rate		x	x	x					
% Substrate disturbed						x			
Temperature[1,2]	x			x					
Water transport/currents				x					

Notes:
[1] Refers to parameter included in model system (see Fulton *et al*. 2005)
[2] Refers to parameters used in the MarLIN sensitivity approach (www.marlin.ac.uk/habitats.php).

Table 27c Chemical parameters / attributes used in prior studies of sensitivity

Chemical	**Biogenic**	**Sand**	**Mud**	**Mixed**	**Seagrass**	**Faunal Turfs**	**Macroalgae**	**Saltmarsh**	**Slow growing epifauna**
Dissolved inorganic nitrogen[1]				x					
Rate of nitrification and denitrification[1]				x					
Redox profile				x					
Salinity[2]	x			x					

Notes:
[1] Refers to parameter included in model system (see Fulton *et al*. 2005)
[2] Refers to parameters used in the MarLIN sensitivity approach (www.marlin.ac.uk/habitats.php).

Table 27d Biological parameters / attributes used in prior studies of sensitivity

Biological	Biogenic	Sand	Mud	Mixed	Seagrass	Faunal Turfs	Macroalgae	Saltmarsh	Slow growing epifauna
Community descriptors									
Assemblage biodiversity (specific phyla)									x
Assemblage structure (specific phyla)									x
Average trophic level[1]				x					
Biomass	x	x	x	x	x				
Density of taxa						x			
Detrital dominance[1]				x					
Diversity	x	x	x	x					x
Environmental position[2]	x	x	x	x	x	x	x	x	x
Functional Group[2]	x	x	x	x					
Mean density of taxa		x							
Net Primary Production (NPP) [1]				x					
Niche breadth			x						
Population density	x	x	x	x					
Production		x	x	x					
Proportion of stock that are juveniles[1]				x					
Species composition		x							
System omnivory index (SOI) [1]				x					
Total system throughput[1]				x					
Trophic efficiency[1]				x					
Waste production[1]				x					
Species descriptors									
Abundance[2]	x	x	x	x	x	x	x	x	x
Abundance and density of associated fauna					x				
Abundance by size class	x	x		x					
Adult/colony size range[2]	x	x	x	x	x	x	x	x	x
Age of sexual maturity[2]	x	x	x	x	x	x	x	x	x
Average size				x					
Bird activity (footprints per area)		x	x						
Body flexibility[2]	x	x	x	x	x	x	x	x	x

Biological	Biogenic	Sand	Mud	Mixed	Seagrass	Faunal Turfs	Macroalgae	Saltmarsh	Slow growing epifauna
Body size distribution			x	x					
Density of post settlement juveniles	x								
Eco-trophic efficiency[1]				x					
Feeding type[2]	x	x	x	x	x	x	x	x	x
Food[2]	x	x	x	x	x	x	x	x	x
Fragility[2]	x	x	x	x	x	x	x	x	x
Growth form/rate[2]	x	x	x	x	x	x	x	x	x
Haul out ground				x					
Immobile biological features				x					
Larval supply/settlement period[2]	x	x	x	x	x	x	x	x	x
Leaf area index (LAI)					x				
Leaf width					x				
Macrophyte cover				x					
Maximum body length (Lmax)				x					
Mobile biological features				x					
Motility[2]	x	x		x					
No. of leaves					x				
Patch size					x				
Phenology								x	
Photosynthetic pathway								x	
Porosity				x					
Potential biological removals (PBR)[1]				x					
Seabird feeding ground				x					
Size (maximum wet weight)				x					
Size class		x							
Shoot density (n/m^2)					x				
Shoot height (cm)					x				
Shoot number					x				
Sociability[2]	x	x	x	x	x	x	x	x	x
Species range and distribution[2]	x	x	x	x	x	x	x	x	x
Species substratum preferences[2]	x	x	x	x	x	x	x	x	x

Biological	**Biogenic**	**Sand**	**Mud**	**Mixed**	**Seagrass**	**Faunal Turfs**	**Macroalgae**	**Saltmarsh**	**Slow growing epifauna**
Tube growth	x								
Reproductive / life history descriptors[2]									
Dispersal potential	x	x	x	x	x	x	x	x	x
Fecundity	x	x	x	x	x	x	x	x	x
Frequency of reproduction	x	x	x	x	x	x	x	x	x
Generation time	x	x	x	x	x	x	x	x	x
Longevity	x	x	x	x	x	x	x	x	x
Recovery rate	x	x	x	x	x	x	x	x	x
Recruitment	x	x	x	x	x	x	x	x	x
Season of reproduction	x	x	x	x	x	x	x	x	x
Size at maturity	x	x	x	x	x	x	x	x	x

Notes:
[1] Refers to parameter included in model system (see Fulton *et al*. 2005)
[2] Refers to parameters used in the MarLIN sensitivity approach (www.marlin.ac.uk/habitats.php).

The most commonly used parameters across all habitat groups were the biological attributes used in the MarLIN sensitivity approach (Table 27). These included abundance, adult/colony size range, body flexibility, fragility, feeding type, motility and reproductive attributes. Specific parameters could not be found in published papers for the chalk reefs habitat group and vertical surfaces habitat group and so they were omitted from the table. Although parameters could not be found, MarLIN has assessed sensitivity for key species occurring in these habitats (www.marlin.ac.uk/habitats.php).

Fishing effort, gear type and total mortality were the most commonly used parameters within descriptors of the activity, and physical parameters, grain-size analysis and sediment composition were consistently looked at (Table 27). Chemical parameters were scarcely used, reflecting the emphasis of physical and biological effects of fishing on community structure.

Numerous parameters are relevant to the description of morphological impacts from fishing activities. These included the physical description of the gear type, the parameters describing habitat modification (e.g. substratum type, particle size distributions, and rate of habitat recovery), the physical attributes of the biogenic habitats (e.g. raised reef and/or habitat creation), and the physical attributes of the species or species group (e.g. epifaunal vs. infaunal, growth height above substratum surface, body size, flexibility and/or fragility), together with parameters that describe the rate at which individuals and populations of species and their communities take to recover their prior abundance, extent or ecological complexity (e.g. community parameters and life history characteristics).

4.2 Effectiveness of parameters for assessing sensitivity

This study shows the wealth of parameters that have been used to examine sensitivity to fishing activities. The multitude of parameters reflects, in part, the different approaches taken, the different purposes for which the studies were undertaken and the availability of information. Natural variability and the range and complexity of responses to impacts have also required a range of parameters to be used for assessment.

Rochet and Trenkel (2003) reviewed the applicability of several parameters to the assessment of the sensitivity of fish communities. They evaluated a variety of parameters based on four important criteria: meaning; expected effect of fishing; exclusiveness; and measurability. They concluded that the most operational descriptors are those that apply to populations and that community and ecosystem descriptors need further development. Although a large variety of indicators have been developed as demonstrated in their review, few have been validated, very few have associated reference points, and still fewer could be delivered to managers for use in decision making.

Body-size distribution (size spectra) has been suggested as a useful descriptor of exploited communities. It is particularly relevant to fishing, which is always size-selective, so it would be expected that fishing should remove larger fish thereby releasing smaller fish from predation (Shin et al. 2005). Size-based descriptors can consider a range of fisheries impacts rather than simply just target species. Rice and Gislason (1996) simulated a multi species virtual population analysis to show that fishing should lead to a decreasing average length of individuals caught. Bianchi et al. (2000) came to the same conclusion. Hiddink et al. (2006) suggested that size-based models can be used to predict large-scale patterns in biomass, production, and species richness of benthic invertebrate communities. However, methodological constraints based on many simplifications and assumptions make it difficult to determine precisely how fishing affects size spectra (Rochet and Trenkel 2003), and as with many other descriptors, size spectra is not strictly specific to fishing impacts (Shin et al., 2005).

Community indices such as diversity have been evaluated as indicators of fishing effects in a number of studies (Ball *et al*., 2000; Collie *et al*., 1997, 2000; Kenchington *et al*., 2001; McConnaughey *et al*., 2000; Schratzberger and Jennings 2002; Thrush *et al*., 1998; Tuck et al., 1998; Veale et al., 2000). Diversity is a description of species richness and evenness (dominance). The rationale behind using diversity indices to measure the effects of fishing is that ecosystems have emergent properties (e.g., intrinsic diversity) which can be altered through unequal removals of target and non-target species (Rice 2000). Jennings and Kaiser (1998) argued that fishing should reduce diversity by selectively removing species. However, diversity has been reported to increase under fishing pressure due to the increases in the evenness index when fishing reduced dominance in an assemblage by decreasing the most abundant stocks (Bianchi et al. 2000). Diversity indices are often difficult to interpret or predict though, as they tend to treat all species as equally informative, and given the selectivity of fishing gear, this is unlikely in reality (Rice, 2000). Species richness is also very difficult to be measured in most marine environments (Rochet and Trenkel 2003).

Biological traits analysis considers a range of biological taxon characteristics to assess how functioning varies between assemblages (Bremner et al. 2003, Tillin et al. 2006). Tillin et al. (2006) analysed the relationship between life history and functional roles within the ecosystem in response to trawling intensity using multivariate analyses. Traits considered included those used in the MarLIN sensitivity approach (age of sexual maturity, feeding type, food, mobility, longevity, reproduction, size etc.). Approaches such as these (Tillin et al. 2006, de Juan et al. 2009, Tyler-Walters et al.

2009) using ecological function and life history descriptors, are a useful way to measure changes in ecosystem function in response to anthropogenic impacts such as fishing (Tillin *et al*., 2006). Jennings *et al*. (1998) used abundance, fishing mortality and life history data in order to examine the effects of fisheries exploitation. Their results indicated that a suite of biological traits determine the response to exploitation but such responses cannot be compared without first accounting for phylogenetic relationships among taxa. The use of total mortality has also been put forward as a strong descriptor as it has a clear meaning and predictable effects of fishing, including reference points (Rochet and Trenkel 2003).

Trait based parameters may be a promising way forward because the effects of fishing on many traits can be predicted (Rochet and Trenkel 2003). However, such parameters have a high data demand, which is often unavailable (Jennings *et al*., 1998) and no single trait can be said to be exclusive to fishing effects (Rochet and Trenkel 2003). Such analyses highlight the need for more accurate biological and ecological information on species in order to inform management decisions (Tillin et al. 2006). In addition, traits alone do not always predict effect or recoverability accurately. Tyler-Walters et al. (2009) found that in most cases fishing sensitivity assessments based on traits alone agreed with assessments based on direct evidence but in some key instances disagreed. For example, while the horse mussel *Modiolus modiolus* is long lived, has a high fecundity, produces large numbers of pelagic larvae with a high dispersal potential, their recruitment is sporadic and poor. One population in particular was thought to be senescent, experiencing little or no recruitment in decades (Comely 1978).

It has been suggested that ecosystem representations (models) are required in order to describe the biomass flows between the different elements of exploited ecosystems and to provide predictive answers to fishing management questions that cannot be provided through real world studies (Pauly et al. 2000).

Fulton et al. (2005) used the ‘Atlantis’ simulation model to evaluate the performance of a suite of ecological parameters covering species, assemblages, habitats and ecosystems in response to the effects of fishing gear as well as broad-scale pressures (e.g., increased nutrient loads). Their results suggest that community descriptors such as biomass, diversity, production and size structure are the most reliable in detecting the impacts of fishing. Pauly *et al*. (2000) used the ‘mass-balance’ model ‘Ecopath’ to simulate the ecosystem impact of fisheries. The model serves to predict changes in biomasses and trophic interactions through time and space. The authors concluded that ‘Ecopath’ offers promise as a tool for evaluating impacts of fishing on ecosystems but it is not fully capable of representing the trophic flows associated with many large aquatic species. Certain draw-backs of model-based approaches include the suggestion that such models rely on unverified assumptions and require extensive data which can be unreliable (Rochet and Trenkel, 2003). Models do not capture the real world perfectly, and values are based on numerous assumptions and relationships that all have a degree of uncertainty (Hiddink *et al.*, 2006). Further development is needed before such models can be used to evaluate the effects of large marine fisheries (Rice, 2000).

Extensive research looking at a variety of parameters has concluded that no single descriptor or parameter can effectively or reliably explain the impact of fishing on community structure and habitat response (Rice, 2000). It has been suggested however, that instead of using a single parameter to measure habitat response, a carefully selected suite of attributes/descriptors would be more useful to encapsulate the effects of fishing (Fulton *et al*., 2005). However, it is still difficult to evaluate descriptors with a known level of rigour and to interpret the results with a high degree of scientific objectivity (Rice, 2000).

4.3 Evidence and Expert Judgement

Evidence is used in this context to refer to the large body of literature regarding species biology, habitat ecology and community interactions that can be used to support an inform sensitivity assessment. Much of this ‘evidence’ is not necessarily parameterised, and or designed to look at the questions of impact or recoverability, resistance or resilience. For example, colonization experiments for artificial reefs provide a wealth of information regarding recruitment and succession, while studies of behaviour and genetics provide information on population subdivision, dispersal and hence recovery. Expert judgement is in part dependant on this body of evidence but at the same time, experts provide an overview of this evidence base.

Several of the approaches to sensitivity assessment are designed to exploit the evidence base and expert judgement, either alone or in combination with specified parameters or traits (see Section 5).

For example, the CCW Fisheries Sensitivity Assessment protocol (Hall et al., 2008) and the Robinson et al. (2008) approach use expert judgement based on the ‘evidence base’ in combination with expert panels to assess sensitivity. The MarLIN approach reviews the evidence base and specific traits in order to reach an assessment (Hiscock and Tyler-Walters 2006, Tyler-Walters et al. 2009). MarLIN gives precedence to the evidence base over traits alone, for the reasons expounded for *Modiolus modiolus* above. Both approaches involve a systematic methodology for the collation and interpretation of traits and evidence.

5 Fishing intensity and impact

The effects of fishing on the marine environment have long been a concern. There is a large and growing number of scientific studies focussing on the effects of fishing on benthic habitats (Bergman and Hup 1992, Currie and Parry 1996, MacDonald et al. 1996, Kaiser 1998, Kaiser et al. 1998, Kaiser et al. 1999, Collie et al. 2000, Frid et al. 2000, Jenkins et al. 2001, Kaiser et al. 2002, Kaiser et al. 2006). Key reviews (Jennings and Kaiser 1998, Sewell and Hiscock 2005, Sewell et al. 2007) on the impact of fishing on marine ecosystems have provided a clear understanding of the direct and indirect effects that fishing activity may have. Publications by MarLIN (Sewell and Hiscock, 2005; Sewell *et al*., 2007; Tyler-Walters *et al*., 2009) and the UK Marine Special Areas of Conversation (SACs) project (e.g., Davison and Hughes 1998, Holt et al. 1998, Jones 2000) have further focussed the effects of fishing toward key species, biotopes and habitats.

5.1 Effects of gear type on habitats

Fishing techniques and equipment have been developed to exploit the behaviour and habitat preferences of target species and to achieve the maximum catch-per-unit-effort (Jennings and Kaiser, 1998). The effect of fishing effort and gear type on benthic habitats has been looked at in a number of published papers (Appendix 2). The majority of parameters looked at for the basis of this report (Table 28) were found in studies conducted on mixed sediments (e.g., Collie et al. 2000; Daan and Gislason 2005; Jennings *et al*., 1998; Kaiser *et al*., 1998), suggesting that this habitat group has been most widely studied with regard to fishing impacts. Many individual empirical studies to date have reported inconsistent findings, and while reviews provide useful summaries of available data, they are often open to interpretation and distortion by different user groups (Kaiser et al. 2006). It has been argued that experimental manipulations of fishing disturbance at the relevant scales are time-consuming and expensive to undertake (Collie et al. 2000).

Meta-analysis is the quantitative summary of multiple, independent studies to detect general relationships permitting ecological questions to be examined over a much wider scale than would otherwise be possible (Kaiser et al. 2006). Collie et al. (2000) undertook a meta-analysis of 39 published fishing impact studies looking at patterns in responses of biota in relation to depth, habitat, disturbance type, and among taxa. The analysis looked at the effects of a one-off fishing disturbance (e.g., a single pass of a dredge or trawl) and results showed that the immediate effect of fishing was to remove about half the individuals. The magnitude of response however, differed significantly with gear type, habitat and among taxa. Inter-tidal dredging and scallop dredging had the greatest initial effects on benthic biota, whereas beam trawling was less significant. The habitats most severely affected were stable gravel, mud and biogenic habitats, compared with less consolidated coarse sediments dominated by opportunistic species, where recovery rate appeared most rapid. The authors concluded that areas that are fished more than three times a year are likely to be maintained in a permanently altered state.

MacDonald *et al*. (1996) developed a sensitivity index to measure disturbance of benthic species in relation to fishing gear type. As with Collie et al. (2000), MacDonald et al. (1996) considered physical disturbance in the context of a single encounter with fishing gear, but followed by a recovery period during which there was no fishing. Mobile fishing gears (trawl, dredge) had a much greater impact over a wider area than static gears (pots, gill nets etc.) which reported low level, localized impacts. The authors concluded that fragile, slow recruiting animals are considered to be most susceptible to disturbance, while fast growing species with good recruitment are less sensitive. With increased levels of disturbance, this will lead to a shift in species composition. Although the study provided the main factors determining likely sensitivity, it only addressed a limited range of biological traits for a limited number of species (Hiscock *et al*., 1999).

Kaiser *et al*. (2006) completed a global meta-analysis of 55 publications which were classified with respect to the following parameters:

- gear type
- disturbance regime
- water depth (m)
- minimum dimension of scale of disturbance (e.g. width of trawl)
- habitat type (mud, muddy sand, sand, gravel and biogenic habitat)
- taxonomic grouping (e.g. by phylum)

Results found that the direct effects of different types of fishing gear were strongly habitat-specific. The most significant impact occurred in biogenic habitats in response to scallop-dredging. Deposit-feeders and suspension-feeders were vulnerable to scallop dredging across gravel, sand and mud habitats. An interesting response was that of soft-sediment biota, which had a predicted recovery time measured in years. Slow-growing large-biomass epifaunal species (e.g. corals, sponges) took the longest time to recover (up to eight years).

Meta-analyses such as these provide important steps forward in understanding and predicting the direct effects of fishing on benthic habitats, which could not be achieved using single studies alone. Results give a possible basis for predicting the outcome of the use of different fishing gears in a variety of habitats. However, the lack of usable data from published studies, especially with regard to recovery time, hinders meaningful interpretation in meta-analysis.

5.2 Long-term responses to fishing

Several causes for long-term changes in benthic communities of the North Sea have been proposed, including the impacts of towed fishing gears (Frid et al. 2000). Rogers and Ellis (2000) examined one of the earliest datasets from research surveys around Britain between 1901-1907 and compared it with more recent data between 1989-1997, in an attempt to compare catch rates of demersal species and to identify any changes in the demersal fish fauna, including non-target fauna, that may have occurred. Increases in abundance of smaller fish strongly suggested the influence of fisheries. Although the results could reflect the increase in capture efficiency through the development of new fishing gears, length-frequency distributions for target species, non-target species and elasmobranchs showed consistent declines in favour of smaller individuals, suggesting the long-term effect of commercial fisheries (Rogers and Ellis, 2000).

Low levels of fishing effort may have significant effects on the diversity and structure of fish communities, but the greatest effects are seen when a previously unfished area is fished for the first time (Jennings and Kaiser, 1998). Frid et al. (2000) also compared historical information to provide a long-term data set of changes in the marine benthos of five selected fishing grounds over sixty years. The authors stated that in three of five areas there was a definite shift in the composition of the benthos due to the increase in catching power of the fishing fleet. Unlike other studies (e.g. Thrush et al. 1995; Kaiser et al., 1997; Tuck et al. 1998), Frid et al. (2000) did not report changes in composition driven by the disappearance of sensitive taxa or the increase in opportunistic taxa. In their conclusion, the lack of control areas or data on fishing intensity made it impossible to link observed changes directly with fishing, but the timing and prevalence of the changes implied that a fisheries link exists.

One of the most reliable ways of obtaining fishing frequency data is from the European Commission (EC) vessel monitoring system (VMS, Eastwood et al. 2007). All vessels operating in EC waters that are >18 m in length are required to transmit automatically their location at a minimum of two hour intervals. Eastwood et al. (2007) used VMS data to locate areas of fishing and assess impacts caused by beam trawlers, otter trawlers, and shellfish dredgers. Estimates of the spatial extent of trawling were based on straight line distances between consecutive positions. Results indicated that demersal trawling affected 5.4 per cent of the seabed.

In many shelf seas, fishing intensity is very high and most fishable grounds will be impacted at intervals of less than one a year (Jennings and Kaiser, 1998). Rijnsdorp et al. (1998) analysed the spatial distribution of fishing effort in a sample of 25 Dutch commercial beam trawlers fishing for sole and plaice between 1993 and 1996 using EC-logbook data and an automated recording system with an accuracy of about 0.1 nm. For an area trawled more than once a year, the impact of beam trawling is a function of the overlap in distribution between beam trawl effort and organisms, both horizontally and vertically, and depends on the fragility of the organisms considered. Deriving estimates of the spatial extent of fishing with demersal gear is problematic as VMS coverage is limited to larger vessels that tend to operate offshore, meaning estimates will often be much lower than what is actually occurring (Eastwood *et al*., 2007). A further problem is in applying generic rules to differentiate between fishing and non-fishing locations, which can introduce errors and reduce accuracy when estimating the spatial extent of fishing (Mills et al. 2006).

Many fishing techniques have direct effects on marine habitats and benthic fauna. A review by Auster and Langton 1999) stated that fishing, using a wide range of gear, produces measurable impacts. However, most studies conducted at small spatial scales make it difficult to apply such information at the regional level where predictive capabilities may allow fisheries management at an ecosystem scale (Jennings and Kaiser, 1998). The development of VMS, fishing impact models and meta-analysis (Collie *et al*., 2000; Frid *et al*., 2000; Kaiser *et al*., 2006) may all hold promise for future predictions of fishing impacts and habitat response, but much more information and data still needs to be made available, especially regarding the biology and recoverability of species and habitats.

5.3 The Importance of Thresholds

Sensitivity assessment requires ‘thresholds’ because resistance (or intolerance) and hence sensitivity are not ‘absolute’ but ‘relative’ terms. Resistance (and hence sensitivity), is dependent on the degree of the impact or effect and the nature of the impact or effect. The degree of impact is usually expressed in terms of the magnitude, extent or scale, duration and frequency of that effect. The nature of the impact is a description of the type of the effector, for example, a physical impact, chemical pollution or biological change.

In many cases, resilience (recoverability) is also relative at is depends in part on the nature of the effect, its duration and frequency (during which recovery is reduced or prevented), and the degree of damage or effect from which the habitat or species population needs to recover.

Sensitivity assessment protocols use a variety of thresholds, categories and ranks. These are:

i. standard categories of human activities and natural events, and their resultant 'pressures' on the environment.

ii. descriptors of the nature of the pressure (i.e. type of pressure, e.g. temperature change, physical disturbance or oxygen depletion).

iii. descriptors of the pressure (e.g. magnitude, extent, duration and frequency of the effect);

iv. descriptors of resultant change (i.e. proportion of species population lost, area of habitat lost/damaged);

v. categories or ranks of recoverability (resilience) thought to be significant; and

vi. resultant categories or ranks of sensitivity and/or vulnerability.

The aim of this standardisation is to ensure that the assessments of 'relative' sensitivity compare 'like with like'.

Categories of Human Activities and Natural Events

Human activities and natural events have been categorised and carefully defined within Environmental Impact Assessments (EIA) since the late sixties. However, in the UK marine environment, the most definitive list was developed by the UK conservation agencies in the Marine Conservation Handbook (Eno 1991) and applied to the Habitats Directive and the Marine Nature Conservation Review (MNCR) by JNCC, and to sensitivity assessment by MarLIN. This list categorised all the activities likely to occur in the marine environment, and was linked to their likely effects on the marine environment via the 'environmental factors' they were likely to affect (see McMath et al. 2000; Tyler-Walters et al. 2001). This list of human activities has continued to be refined under the Water Framework Directive (WFD), the UK Marine Monitoring Assessment Strategy (UKMMAS) and for the 'Charting Progress 2' (CP2) review of the status of the UK's marine environment (in prep.). Robinson et al. (2008) used the standard list of activities and pressures developed under CP2 in their sensitivity assessment procedure.

Nature of the Pressure

In this review, we are only concerned with fishing impacts. Nevertheless, fishing and shell fishing are diverse activities, using a range of different techniques and equipment at different depths, on a variety of substrata, in both inshore and offshore waters. Most studies have categorised the nature of the pressure by method or gear type, e.g. beam trawl, otter trawl, or scallop dredge. The most extensive review of fishing activities (including those associated with shellfisheries) was conducted by CCW and Hall et al. (2008). Hall et al. (2008) list a total of 38 different fishing activities and gear types, divided into 12 separate categories. Access to fishing areas across the intertidal was added as another category (Tyler-Walters and Arnold, 2008).

Degree of Pressure

MarLIN chose ‘benchmark’ or ‘threshold’ levels of effect for 24 separate environmental factors or pressures based on a short review of the likely effects of marine activities and the level of change in any given pressure likely to result in an effect of marine species or habitats. In each case the magnitude of the change, and its duration were specified. For fishing impacts, a single pass of a bottom trawl, such as scallop dredge was chosen as the benchmark. This was adopted based on the work of MacDonald et al. (1996). Recent studies (e.g. Kaiser et al. 2006) have also suggested that most damage from bottom trawling gear occurs in the first occurrence, suggesting that a single pass is a sensible threshold, especially for habitats dominated by long-lived species and biogenic habitats.

Hall et al. (2008) developed a set of different intensities of impacts for each of the different gear types, ranging from high to low; the exact intensity (in terms of number of trawls per unit area and per unit time) depended on the type of gear. In addition, they used a ‘single pass’ as a separate intensity. The emphasis of their study was on capturing levels of intensity for each gear type that corresponded to those used in practice by fishermen.

Few studies have used trawling intensity derived from VMS data to determine sensitivity. Hiddink et al. (2007) examined the effects of different intensities of trawling on the sensitivity of marine habitats, based on a model of sensitivity that assumed that sensitivity of a marine habitat was related to the recovery of species biomass and productivity. They used a range of trawling intensities from zero to five per year in their model but did not use thresholds of intensity to determine sensitivity.

Descriptors of Change and Assessment Scales

All sensitivity assessment protocols develop scales or series of ranks against which to assess resistance (tolerance), resilience (recoverability), sensitivity and vulnerability. In most cases, these scales reflect the reasons the approach was designed, the audience they are designed to communicate to, and the ‘desirable’ state, conservation or management objectives served by the assessment.

For example, MarLIN uses a broad scale to assess intolerance (resistance) based on the degree of damage that is likely to result from an effect (see section 6). In short, the scale varies from ‘significant damage, most of the population destroyed’, through ‘a proportion of the population destroyed or removed’ to ‘only sublethal effects on the population’. The scale was chosen because it reflected the likely levels of damage but also because it could be applied to the evidence base. However, Robinson et al. (2009) based their scales on the levels outlined for definition of the OSPAR list of threatened and declining species (OSPAR Annex V, OSPAR Commission 2008). The OSPAR scales provide a basis for reporting within UKMMAS and Charting Progress 2, for which their methodology was designed.

In recent empirical studies, e.g. Kaiser et al. (2006) recovery was defined to have occurred when the regression fits to the data returned to with 20 per cent of the original population abundance.

It is important to set scales against which to rank resistance (tolerance), resilience (recoverability), sensitivity and vulnerability, that are meaningful to the intended audience, are relevant to the management objectives being addressed, and that can be used with the evidence base and/or approach used for assessment.

6 Sensitivity assessment

The approaches developed to assess sensitivity in the marine environment vary depending on; the aims of the study, the type of impact or activity examined; the ecological level considered (species, community, biome etc.), the geographical or spatial extent of the study and the information available.

Methods for assessing sensitivity principally vary in the scale of assessment and this influences the type of attributes that are chosen to represent sensitivity in the assessment.

Sensitivity assessments can be categorised into three broad groups:

- those that primarily assess the sensitivity of selected species;
- those that are primarily used to assess the sensitivity of biotopes and habitats; and
- those that are used on a regional/broader scale and are based on high level environmental characteristics.

These approaches are described in general terms below and representative approaches of assessing sensitivity are described for each group. For the purposes of the WFD Commercial Fisheries Risk Assessment, it was recognized that, to assess sensitivity, it was desirable that an approach would enable a consistent assessment to be made across a range of habitat types. This review has therefore focused on types of approaches that have been applied or could be further developed to assess fishery impacts for a range of habitat types.

In essence, the measures of assessing sensitivity at the biotope level, described in this review, are an extension of methods of assessing sensitivity at the species levels. This is logical when it is recognised that the sensitivity of a biotope depends on the sensitivity of the constituent species. It is not possible, or desirable, to assess sensitivity based on every constituent organism, decisions are therefore required on which species should be selected to represent sensitivity. The selection protocols of different approaches (SensMap and MarLIN) are discussed below.

One apparent weakness of using a species / biotope approach to assess sensitivity, is that information on the sensitivity of the habitat is not incorporated (e.g. changes to sediment and substratum). If the habitat suitability is affected, then recovery may not take place, or be delayed, as the location can no longer support the biotope. The regional assessments described below incorporate further attributes that consider the sensitivity of the habitat (among others). Such approaches have been primarily used to assess management planning at regional scales for oil spills. However, this approach could be modified to provide sensitivity assessments for commercial fisheries management.

The final review section on approaches to assessing habitat sensitivity describes examples where sensitivity assessments have been specifically linked to the distribution of spatial pressures to develop vulnerability maps. All of the sensitivity assessments described could be used in vulnerability assessments, so the intention of this section was to provide some notable, recent examples or projects.

Primarily approaches vary in:

- the pressures (impacts) that are included in the assessment;
- decisions on which and how many species to include in the assessment;

- the species traits chosen to represent sensitivity;
- the approach to information gathering (for example, expert judgement vs. literature review);
- the method used to compile or combine information (scoring systems);
- weighting of components within the assessment; and
- the assessment outputs, e.g. ranks, categories, index scores.

6.1 Species Level

Assessing the sensitivity of species to stress or perturbations is the most conceptually straightforward approach and the most widely used. Measures of the impacts of species, for example, have been used to inform toxicity assessments, the management of commercially important species and the conservation of species populations.

In the UK, most broad-scale marine assessments of condition have been based on the sensitivity of sessile or sedentary species including macroalgae and macro-invertebrate species (Tyler-Walters and Hiscock 2005; Tyler-Walters et al. 2009, Hall et al. 2008). In terms of sensitivity assessments, these have a number of desirable characteristics; they are linked to a particular location and, as species vary in their sensitivity to anthropogenic impacts, they can be informative of the factors affecting a habitat over a long-time scale.

There is a wide scientific literature available on the effects of pressures on species and the use of single species populations as indicators of pollution and other types of disturbance. Species may indicate exposure to pressures based on characteristics such as:

- the presence or absence of the species from a location;
- the demography of the population, e.g. a population composed of only old or young species, is informative about long-term habitat suitability; and
- physical characteristics including size and the presence of gross physical damage.

Two types of species sensitivity assessment and/or study were identified as particularly relevant to the purpose of this review and are described in greater detail. These are the Life Form Sensitivity Assessment approach that was developed by (Holt et al. 1995, 1997) and developed further to assess fishing impacts by MacDonald et al. (1996) (Table 28). Biological Traits Analysis also assesses the sensitivity of species based on a number of parameters.

Life Form Sensitivity Assessment

Holt et al. (1997) assessed the sensitivity of three species/life forms (*Zostera* (eelgrass), *Sabellaria spinulosa* reefs and brown algal shrubs, for CCW. This study built on a major scoping study that applied the approach to a range of 'life forms' (Holt et al., 1995) The aim of this work was to evaluate the sensitivities of the selected species/life forms in a general sense and with regard to various impacts. This approach has been further developed (as discussed below) to assess fishery impacts and has underpinned development of widely used approaches, e.g. the MarLIN approach.

Table 28 Examples of approaches to assessing the sensitivity of single species and the applications of these assessments

Metrics	Examples	References
Traits Analysis Assessing recoverability of communities to fishing.	Uses life history traits to assess recoverability based on recruitment and growth.	Bremner et al. 2006b, Tillin et al. 2006, Bremner 2008
General Sensitivity Assessment	Selected key indicator or sensitive species or species groups. Uses life history traits to assess sensitivity for five criteria (longevity, fragility, stability, intolerance and recoverability). Considers information on other organisms within the community.	Holt et al. (1995)
Life Form Sensitivity Approach	Sensitivity of key species assessed based on resistance and recoverability to an impacting activity (fishing disturbance). Uses a simple formula to define sensitivity value.	MacDonald et al. (1996)

This approach is significant in that it has been used to develop further sensitivity assessments but is also relevant to this study as the tables produced provide an example of an audit trail for a sensitivity assessment. Keeping records of the information and categorisation that underpins a sensitivity assessment allows the assessment process to be transparent and would also support assessment updating for new habitat types or information.

In essence, the approach assesses sensitivity based on species traits that represent resistance (intolerance) and recoverability. For the initial study (Holt et al. 1995), the sensitivity concentrated on the selected species/life forms with some consideration of the associated community.

A systematic literature review was used to evaluate sensitivity based on species traits that relate to resistance (intolerance) and recoverability. These were longevity, fragility (based on physical impacts) and population stability (to biological and physical pressures). These categories were scored from 1-5 for single species. The categories for each of these traits are shown in Table 29. Information on responses to a number of impacts was used to assess intolerance and recoverability.

Compiling a matrix in this way provides a general indication of the sensitivity of the life form in consideration and allows comparisons to be made between life forms. These results were not combined any further, e.g. to create an index score or to rank species, to give an overall assessment of sensitivity. This step was taken in later studies as described below (SensMap approach).

Species Sensitivity to Fishing

MacDonald et al. (1996) developed an index of species sensitivity to fishing disturbance, which allowed the sensitivity of species to fishing gears to be ranked. This study considered disturbance only in terms of the physical action of the gear on the seabed and the area over which this action extends.

Table 29 Suggested scoring system for assessing sensitivity on a numerical basis (from Holt et al. 1995)

	1	2	3	4	5
Longevity	Annual or shorter	1-2 years	3-5 years	5-10 years	>10 years
Fragility	Very robust	Fairly robust	Moderately fragile	Fairly fragile	Very fragile
Stability	Characteristics of rapid colonisers / transient communities	Major fluctuations in populations likely every 1-2 years	Major fluctuations in populations likely every 3-5 years	Major fluctuations likely every 5-10 years	Major fluctuations rare.
Intolerance	Very tolerant to a wide range of environmental changes	Tolerant to a moderate variety of environmental changes	Neither tolerant or intolerant	Intolerant to a moderate variety of environmental changes	Very intolerant
Recoverability	Recovery from most damage within a year	Recovery from most damage likely 1-2 years	Recovery from most damage 3-5 years	Recovery from most damage 5-10 years	Recovery from most damage unlikely within 10 years.

Their approach is a useful example of index development and the use of a consistent approach to assessment, showing how traits can be selected and combined in order to assess sensitivity. The approach is highly relevant to this review as it focused on fishing disturbance. The approach also demonstrates how information on the fishing impact and intensity can be categorised, as a measure of impact was incorporated into the assessment.

The assessment of sensitivity was based on species traits relating to tolerance (assessed through fragility), responses and recoverability. The theoretical sensitivity of individual species was assessed on the basis of how well they cope with a **single** encounter with fishing gear and on their likely recovery from destruction in terms of their **reproductive strategies**. Sensitivity was assessed based on species traits relating to the two criteria:

- the fragility of individuals of a species that come into physical contact with the disturbing force (based on the organisms' physiology and/or structure including strength or flexibility). The species traits used included position, circadian rhythms or abilities to avoid and withdraw to evade capture and/or damage.
- the ability of the species to recover to its former population or physical status with the disturbed area (the ability of damaged organisms to repair or regenerate lost or damaged parts, to continue occupying the disturbed habitat, the supply of larvae to the habitat and their settlement success and recruitment of settled larvae to the adult population).

The index of sensitivity is calculated based on the consistent assessment of each species using a formula (Equation 1). The formula weights recovery potential, as this was considered to be the most important component of sensitivity.

$$S = (F \times I)\, e^{R} \qquad \text{(Equation 1)}$$

Where:

S is the sensitivity index

R is recovery and is scored on a scale of 1 to 4, equivalent to short, moderate, long and very long recovery period/no recovery timescales).

F is fragility (scored on a scale of 1 to 3, equivalent to not very fragile, moderately fragile, and very fragile). A later study assessing the impact of pot fishing on benthic species used a 5-point damage scale but suggested that a 3-point scale is more easily used (Eno et al. 2001).

I is the intensity of the impact (scored on a 3-point scale, equivalent to low, moderate and high intensity). Intensity depends on the type of gear and factors such as its momentum and the depth of penetration into the substratum, for example dredges score more highly than a long-line.

e is the base rate of exponential growth, a constant and the base of natural logarithms.

The sensitivity scores were then normalised for species by dividing each score into the maximum possible score using the relationship:

$$S_{norm} = \frac{S_n}{S_{\max}} \times 100 \qquad \text{(Equation 2)}$$

Where S_{norm} is the normalised sensitivity score for any species *n* (i.e. S_n the *nth* value of S) and S_{max} is the maximum possible sensitivity for the most disturbing fishing activity.

Species Selection

MacDonald et al. (1996) provided sensitivity assessments for 35 species, including those that create biogenic habitats i.e. *Sabellaria* reefs, for three levels of gear intensity. Emphasis in the sensitivity assessment was given to species considered to be of key importance and those that structure communities. These include species that provide architectural structure e.g. *Modiolus modiolus*, modify environmental conditions e.g. *Zostera* beds, or those which structure the assemblage through trophic interactions such as major grazers or predators. However as identifying key species may be difficult the authors also suggested the use of indicator species, which were defined as those whose abundance may provide a guide to levels of fishing disturbance and therefore whether communities are natural or altered by fishing. These would be the most sensitive species so again this may skew the results.

This approach was further developed and applied to the assessment of habitats and biotopes and spatial mapping of vulnerability, through the SensMap project (McMath et al. 2000). SensMap involved the application of the protocol to biotopes to assess sensitivity to a broad range of marine activities and to map sensitivity (see Section 5.2 for discussion).

Species Biological Traits Analysis

Biological Traits Analysis (BTA) approaches to assessing ecosystem conditions and species distributions are based on life history and ecological traits expressed by species, e.g. feeding type, longevity, position within habitats. Multivariate approaches to trait analysis were initially developed for freshwater systems (Chevenet et al. 1994, Usseglio-Polatera et al. 2000) and have been used in the marine environment to explore how fishing alters functioning (Bremner et al. 2003, Tillin et al. 2006), to identify how trait composition is linked to environmental gradients (Bremner et al. 2006b) and to assess how function may be used to delineate boundaries of Marine Protected Areas (MPAs) (Bremner et al. 2006a).

A further example of the use of species traits to predict recoverability was developed by Marine Ecological Surveys. They assessed the extent to which the recoverability of aggregate licensed areas (assessed on return to biomass and biodiversity in individual genera and communities) can be predicted based on the species traits present (Marine Ecological Surveys Ltd, see Newell 2006).

The traits chosen to represent the recoverability potential of a species were chosen to encapsulate recruitment and growth information.

- size,
- fecundity,
- life-span,
- age at maturity,
- larval mode and
- adult mobility.

The recoverability study assessed 119 species and derived vulnerability and recoverability scores for each of these in relation to 24 categories distributed between the six traits. The scores for each taxon were scaled out of a possible total of six between vulnerability and recoverability as shown in Table 30 for the category size. From this table it can be seen, for example, that an organism smaller than 1 cm is predicted to have low vulnerability and high recoverability. The score indicates whether the trait being assessed is robust (R) or vulnerable (V).

The analysis allowed the vulnerability of individual taxa to dredging, and the likely time required for colonization, to be estimated. The taxa scores were also used to estimate the time required for restoration of community structure following the growth of the colonising individuals to adult size.

Table 30 Scoring vulnerability and recoverability for the trait category size

Size	Vulnerability	Recoverability	Score
<1 cm	1	5	R
1-3 cm	2	4	R
3-10 cm	3	3	
10-20 cm	4	2	V
>20 cm	5	1	V

6.2 Habitat / Biotope Sensitivity Assessment Methodologies

A number of methods for evaluating the sensitivity of habitats and biotopes have been developed. The pressures considered and the parameters included in the assessment are as equally variable. A separation can be made between studies that have provided information on the impacts of pressures and the development of methodologies that predict sensitivity based on the knowledge gained from these.

The effects of impacts are assessed directly based on field based observations and experiments to produce qualitative and quantitative models. Information gained from these studies informs the assessment of habitat sensitivities

Approaches to predicting sensitivity that can be used in broad scale habitat mapping include multivariable indices that in the past have provided a popular method to assess the sensitivity of habitats and biotopes. A recent development has been the use of information from the scientific literature and expert judgement to predict sensitivity, through the compilation of data tables/matrices.

Examples of different approaches to assessing sensitivity and predicting sensitivity are discussed below and are summarised in Table 31.

Table 31 Summary of Approaches to Sensitivity Assessment based on species and habitats

	Metrics	Description	Examples
Qualitative Models	Qualitative Models	Ranking of habitat sensitivity in relation to an impacting activity, based on observations and expert judgement.	Conceptual models of Auster (1998)
Quantitative (Empirical) Models	Regression Models	Assessment of sensitivity to fishing using meta-analysis approach. Developed quantitative models to assess change in no. of individuals, species richness and effects at the genus and higher taxonomic levels. Variables include gear type, five habitat types (based on sediment/substrate)	Collie et al. (2000).
Multivariable Indices	SENSMAP Approach	Adopted MacDonald et al. (1996) approach. Sensitivity assessment derived from sensitive species. They suggested that for biotopes either, i) report mean sensitivity, or ii) report highest sensitivity.	McMath et al. (2000)
	MarLIN Approach	Sensitivity assessment based on information on key or important species intolerance and recoverability, together with habitat characteristics.	Hiscock and Tyler-Walters (2006).
Matrix Approaches	Beaumaris Approach	Assesses habitat sensitivity to a pressure (fishing), at different intensities, to derive a sensitivity score (High, medium, low). Used a matrix containing three factors; fishing metier, intensity of fishing, and habitat sensitivity.	Hall et al. (2008), Tyler-Walters et al. (2008).

6.2.1 Qualitative Methods

The simplest models that describe the sensitivity of habitats are those that provide qualitative predictions on the effect of impacts. Examples of these are the general models of the impacts of fishing gear in habitats with different structural components (Auster 1998) and the general schema of the vulnerability of different habitat types to fishing disturbance presented by Bax and Williams (2001). Both of the examples considered here incorporate information on morphological impacts of fishing activities on habitats and consider changes across a range of habitat types.

Auster (1998) developed a generalised model of fishing activities on habitats complexity based on empirical observations. Each habitat type was categorised for an unaffected state and scored numerically based on the structural complexity. Values for the habitat structure of highly affected habitats, based on observations, were then incorporated into the model. The model predicted that linear increases in fishing effort would lead to linear decreases in habitat complexity. The model incorporates a range of habitats varying from those with the lowest morphological complexity (bedforms) to the highest (piled boulders).

Sensitivity (defined as vulnerability in the study of south-eastern Australian seabed habitats to fishing) was assessed by Bax and Williams (2001). Habitat vulnerability (sensitivity) was defined as the product of resistance and resilience. Figure 1 shows the conceptual model of vulnerability of different substratum types and incorporates natural levels of disturbance. The model shows that unconsolidated sediments in high energy environments (i.e. where natural disturbance is high) are less sensitive (more resilience) than substrata in low energy environments. While complex rock habitats have higher resistance to fishing disturbance than sedimentary substrata, they have low recovery. Such schematics are useful to categorise conceptually the sensitivity of different types of habitats in ways that can be usefully communicated to stakeholders.

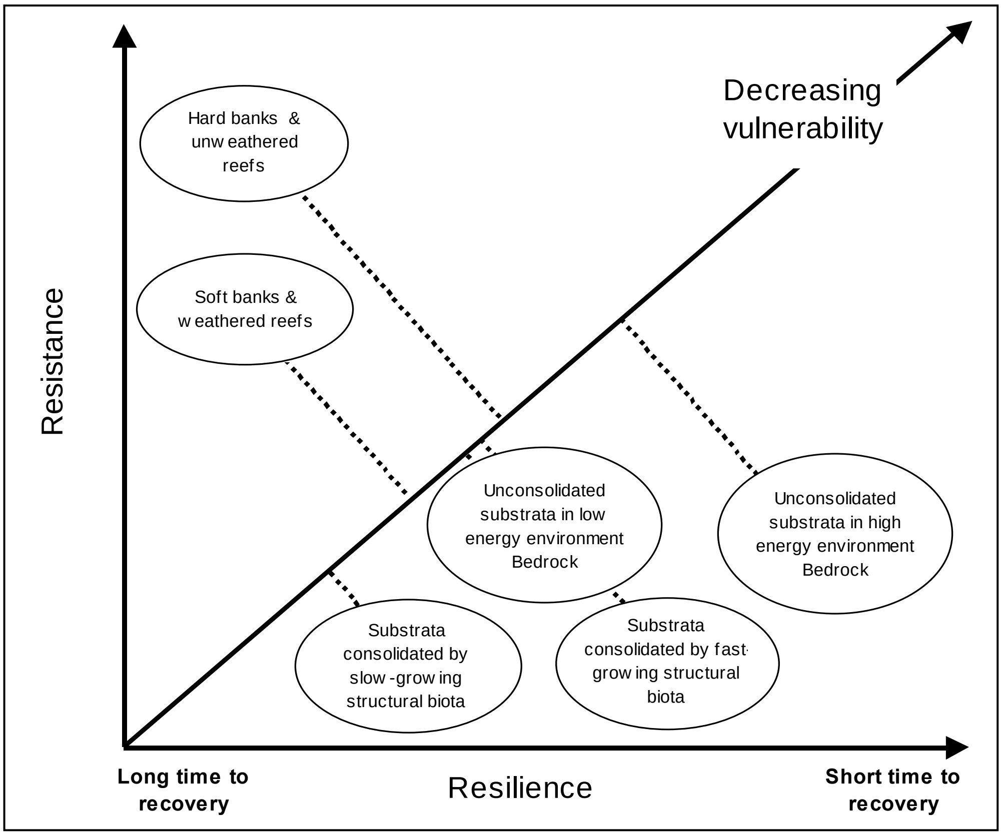

Figure 1 Vulnerability of different substratum types from Bax and Williams (2001)

6.2.2 Quantitative Methods based on Assemblage Characteristics and Properties

Meta-analyses that use information from a number of empirical studies have proved to be a powerful tool to evaluate fishing effects and habitat sensitivity. Collie et al. (2000) used information from a number of studies on the impacts of fishing on benthic habitats to develop quantitative (General Linear Model (GLM) framework, Generalised Additive Models (GAM) and ANOVA) models of the sensitivity of different benthic habitat types. Sensitivity was evaluated based on the effects on the abundance of individuals, species richness and (where possible) effects at genus and higher taxonomic levels.

The study assessed the effects of different gear types, the intensity and scale of fishing effects, region and depth and habitat type. The modelled patterns of recovery showed that the sensitivity of taxa varied by gear type, with inter-tidal dredging having the most negative impact followed by scallop dredging. Recovery patterns also varied between habitat types indicating inter-habitat sensitivity differences to fishing.

A later study by Kaiser et al. (2006) used the opportunity provided by a large increase in the number of published studies on fishing effects, to analyse 101 fishing impact experiments. The study showed that the effects of fishing were habitat and gear type specific. Scallop dredging had the most impact in soft-sediment habitats (mud, sand and gravels, and affected deposit and suspension feeders. Long-lived large organisms took longer to recover than small species with more rapid life-histories.

Empirical models of fishing effects, based on meta-analyses of collected studies have provided useful evidence of fishing impacts and recovery times in different habitats according to gear types (Collie et al. 2000, Kaiser et al. 2006). A weakness encountered with meta-analyses is that they rely on the information available, in this case published studies. However, this approach does provide a synthesis of the available information and can identify trends that can be further explored.

6.2.3 Multivariable Indices for deriving sensitivity assessments

SENSMAP

Inshore marine biotopes (intertidal and subtidal) in the Irish Sea were mapped and the sensitivity of these to a range of maritime activities was assessed using a methodology based on the approach developed by MacDonald et al. (1996). These data were combined to produce broad scale maps of sensitivity for the Irish Sea (McMath et al. 2000).

The approach demonstrates a method whereby species sensitivity assessments can be combined consistently to assess the sensitivity of biotopes, biotope complexes or life form levels. A sensitivity index score is developed and the scores are categorised to develop regional maps of sensitivity. Aspects of the approach, including species selection and confidence rating, discussed below are examples of how consistency and transparency can be incorporated into the assessment. The approach also developed a method of assessing additive impacts, which may be useful for fishing impact classification, where more than one effect might be considered, e.g. removal of habitat forming species and disturbance to the sediment, or where different types of fishing gears were considered.

SENSMAP Methodology

Following the MacDonald et al (1996) methodology, the sensitivity of a species was measured by its initial intolerance to a pressure (degree of resistance) and its ability to recover from the impact (Table 32). The sensitivity of a feature is based on an analysis of the sensitivity of species present to the pressures associated with activities. Subjective scores were assigned based on expert judgement and information from the scientific literature; these scores were ranked to produce the sensitivity maps.

Table 32 Characterisation of species found in biotopes (from SensMap report)

Species within biotopes	Description
Highly sensitive	Useful as a warning of the existence of low-level detrimental effects but does not generally provide an accurate representation of the overall biotope sensitivity.
Rare and Scarce	Important from a conservation perspective but cannot always represent the overall biotope sensitivity reliably, as the proportion of the biotope which they compose can range from substantial to negligible. Some may be at the edge of their ecological range and so highly sensitive
Characterising species	Those that best characterise each particular biotope. Using those of the greatest biomass would provide a more representative assessment of the sensitivity of the major biotic components of the biotope.
Keystone species	Those that if lost from the biotope would spell both serious change in the community's species composition and long term survival. Keystone structural species provide a distinct habitat that supports an associated community. Keystone functional species maintain community structure and function through interactions with other members of that community (for example, predation or grazing). Loss/degradation of these species would result in rapid, cascading changes in the community.

The procedure used the following stages to derive sensitivity information and to display in a mapped format (Steps iii – iv are discussed).

i. Selection of an activity

ii. Selection of associated pressures at particular intensities

iii. Selection of a species, biotope or life form etc.

iv. Sensitivity value calculated from **species intolerance and recovery scores**

v. Sensitivity values **ranked** into groups, which are then represented by different colours for sensitivity spatial mapping purposes.

Step iii) Selection of Species, Biotope or Life Form.

Biotopes consist of a number of species, which the SensMap report (McMath et al. 2000) suggested could be classified as in Table 32. Characterising species and keystone species were identified as being most useful to form the basis of sensitivity assessments, as the loss of these species will alter the character of the biotope.

The number of species that will be selected to form the basis of an assessment will depend on factors such as practicality, the number required to represent sensitivity adequately and the number of keystone/important species identified within the biotope. The authors suggested that three species are chosen to derive a sensitivity assessment. The contribution of a keystone species is weighted by a factor of two in the assessment to account for its importance.

Biotope sensitivity is based on the sum of sensitivities of three component species. Biotope sensitivity is ranked based on the cut-off points used in sensitivity species assessments and the values are normalised to fall between 0-100 to represent biotope sensitivity.

Step iv)

The SensMap project modified the method of MacDonald et al. (1996) to assess the sensitivity of species that respond to different factors. The assessment depends on determining **intolerance (I)** and **recovery (R**) for a range of pressures and pressure intensities and species recovery. The components used to assign the intolerance and recovery scores are shown above in Table 33. Guidance and examples of sensitivity assessments for species are provided in McMath et al. (2000).

Table 33 Intolerance and Recovery components used to assign scores

Intolerance (I) Components	**Attributes**
Exposure	Mobility, feeding mechanisms, habitat preferences, and growth structure
Fragility	Direct mortality, effects on growth and reproduction, behavioural responses and indirect effects (through effects on other biota) development stage e.g. larval stage or adult.
Season of activity or factor	If species' energy reserves are depleted, species intolerance may increase; seasonal migratory patterns of mobile species may influence species susceptibility to factors.
Original condition of species	If species are stressed before being exposed to a factor, species intolerance may be increased. For example, species occurring at the limits of their geographical range are often less capable of handling additional stresses.
Recovery (R) Components	
Recruitment	Frequency/length of reproduction season, development mechanism, age at maturity, growth rate, reproductive type, distribution, influence of pressure on recruitment success.
Recolonization	Mobility of adults, species distribution, influence of pressure.
Regeneration	Regeneration and regenerative growth rate, influence of pressure.

Intolerance and recovery, were scored between 1 and 100 and combined to produce the sensitivity score (equation below). Sensitivity scores were then grouped into bands and each of the five categories were assigned a colour label for mapping purposes.

A score between 1 and 100 was assigned for each of the three recovery components (recruitment, recolonization and regeneration). These scores were then weighted in the ratio 8 (recruitment): 1 (recolonization): 1 (regeneration) to reflect the relative importance of recruitment to recovery.

The modified index from MacDonald et al. (1996) used to derive the sensitivity score is:

$$\mathbf{S = I \times R^2} \qquad \text{(Equation 3)}$$

Where:

S = sensitivity of a species to a factor,

I = intolerance of a species to a factor at a particular intensity, and

R = species recovery.

The sensitivity of a species to multiple simultaneous activities was calculated as:

Species sensitivity multiple factors = [∑ I species to factors] x R^2 (Equation 4)

This assumes that pressures have an additive effect, where pressures have synergistic or antagonistic effects intolerance scores should be assigned based on the combination of factors.

Combining Biotope Sensitivity Assessments

Because all broader units of classification consist of a specific set of biotopes, their sensitivity can be derived from the sensitivity of their component biotopes. Where biotope information exists, then the combined sensitivity could be reported as the mean or highest sensitivity, depending on the study aims.

It is suggested that the sensitivity of the component species should also be represented at mapping stages as the contribution of each species to overall sensitivity is important in determining recovery e.g. where a keystone species has much lower or higher sensitivity to impacts that the other species (McMath et al. 2000).

Confidence Ratings

Application of the SensMap methodology in the Irish Sea highlighted that the information available to support sensitivity assessments varies between species and that sometime the information available has limitations, e.g., extrapolated from laboratory conditions which do not reflect natural conditions. In some cases, the assessment would have to be based on expert judgement. Therefore, confidence assessments were applied to the combined intolerance and recovery assessment, where the assessment was based on the lowest confidence measure. Three categories were used as described in Table 34.

Table 34 Confidence labels for species intolerance and recovery used in the SensMap assessment (McMath et al. 2000)

Confidence Rating	Description
High	Directly relevant information available: assessment is based on direct relevant experimental data or personal observation of species intolerance or recovery.
Moderate	Related information available: assessment is based on extrapolation (e.g. similar species) of related experimental data or personal observation of species intolerance or recovery.
Low	No information on species intolerance or recovery is available: assessment is based on knowledge of biological and ecological requirements of species.

Further development

Hiscock (1999) broadened this approach and identified many of the biological traits important to such assessments. This technique was then adapted in (Hiscock and Tyler-Walters 2006, Tyler-Walters et al. 2001) and forms the basis of the MarLIN approach.

MarLIN Approach

The MarLIN approach assesses the sensitivity of species and biotopes to 24 separate environmental factors (including physical, chemical and biological) by combining a measure of intolerance with a measure of recoverability. Intolerance is defined as the susceptibility of a species population to damage or death from an external factor. Recoverability is defined as the ability of a habitat, community or individual species to redress damage sustained as a result of an external factor.

All the information used in the assessment, and the decisions made are documented, and the results presented as a Biology and Sensitivity Key Information review, and placed in the public domain via the internet.

In the case of biotope, intolerance and recoverability, the steps involved in assessment are:

i. Collate 'key' information for biotope

ii. Select species indicative of biotope sensitivity

iii. Review key information for the selected species

iv. Indicate quality of available data

v. Assess intolerance, recoverability and sensitivity of indicative species to environmental factors (using benchmarks)

vi. Assess overall intolerance and recoverability of the biotope

vii. Assess sensitivity of the biotope

viii. Assess the likely effect of the factor on species richness

ix. Review and place-online and subject to peer review.

x. Revise if required by referee.

Methodology - steps i-iv.

The methodology is underpinned by a review of available literature on the chosen species. The information collated includes life history characteristics, distribution, environmental preferences and any effects of environmental perturbation.

For biotopes the information reviewed includes; biotope classification, ecological relationships, seasonal and longer-term habitat changes, habitat complexity, productivity, recruitment processes, time for community to reach maturity, habitat distribution, species composition, sensitivity and marine natural heritage importance.

The biotope sensitivity assessment is based on selected species and the review is used to select appropriate species for this. Key structural and functional species are preferred (see also SensMap methodology). The presence of particularly threatened or rare species, and the sensitivity of other members of the associated community, their recruitment process and community succession are taken into account in the assessment.

MarLIN uses a six-point scale to indicate the specificity of the information available to support the assessment of sensitivity.

Step V - Assessment of Intolerance and Recoverability

The MarLIN programme developed a set of benchmark levels of environmental change against which to assess sensitivity.

The likely intolerance of a species is assessed with respect to a specified magnitude and duration of change (the benchmark or threshold). Degrees of intolerance are ranked into six categories ranging from 'not relevant' where the species can avoid the factor or is protected from it, to 'high' where the species population is likely to be killed/destroyed by the factor (Table 35)

Table 35 Categories of Biotope Intolerance (from MarLIN approach)

Rank	Definition
High	Keystone/dominant species in the biotope or habitat are likely to be killed/destroyed by the factor under consideration.
Intermediate	The population(s) of keystone/dominant species in a community may be reduced/degraded by the factor under consideration, the habitat may be partially destroyed or the viability of a species population, diversity and function of a community may be reduced.
Low	Keystone/dominant species in a community or the habitat being considered are unlikely to be killed/destroyed by the factor under consideration and the habitat is unlikely to be damaged. However, the viability of a species population or diversity / functionality in a community will be reduced.
Not sensitive	The factor does not have a detectable effect on structure and functioning of a biotope or the survival or viability of keystone/important species
Not sensitive*	The extent or species richness of a biotope may be increased or enhanced by the factor.
Not relevant	Sensitivity may be assessed as not relevant where communities and species are protected or physically removed from the factor (for instance circalittoral communities are unlikely to be affected by increased emergence regime).

The likely recoverability of a species from disturbance or damage is dependent on its ability to regenerate, regrow, recruit or recolonise depending on the extent of damage incurred (the intolerance). Recoverability is only applicable if (and when) the impacting factor is removed or has stopped.

Recoverability is ranked according to eight categories from 'not relevant' where sensitivity is not relevant or cannot be assessed to 'none' (Table 36).

Intolerance is assessed against the defined benchmarks (or thresholds) of effect set for each of 24 separate environmental pressures. Precedence is given to direct evidence of effect in the literature, i.e. that a given pressure effected a population of the species or habitat in question. Similarly, evidence on recolonization rates in similar habitats is also used where available.

Table 36 Categories of Biotope Recoverability (from MarLIN approach)

Rank	Definition
None	Recovery is not possible
Very low / none	Partial recovery is only likely to occur after about 10 years and full recovery may take over 25 years or never occur.
Low	Only partial recovery is likely within 10 years and full recovery is likely to take up to 25 years.
Moderate	Only partial recovery is likely within five years and full recovery is likely to take up to 10 years.
High	Full recovery will occur but will take many months (or more likely years) but should be complete within about five years.
Very high	Full recovery is likely within a few weeks or at most six months.
Immediate	Recovery immediate or within a few days.
Not relevant	For when intolerance is not relevant or cannot be assessed. Recoverability cannot have a value if there is no "intolerance" and is thus "Not relevant".
Insufficient information	Insufficient information

In addition to the direct evidence, and/or where direct evidence is not available, relevant biological traits are used to assess the intolerance and recoverability. The biological traits used depend on the nature of the pressure. Rather than use a formula or equation to combine trait scores, the relevant traits for each pressure are assessed via simple decision flow charts (see Tyler-Walters et al., 1999; 2001), one for each pressure under consideration. Another set of flow charts are used to assess recoverability. The use of flow charts ensures that the information is used in a systematic and transparent way without recourse to formulae.

Step Vii - Sensitivity Assessment

The intolerance and recoverability categories are used to assign a sensitivity value, and are combined based on the sensitivity matrix below (Table 37). This combination of intolerance and recoverability is based on the definition of sensitivity developed for the Review of Marine Nature Conservation (Laffoley et al. 2000). For instance, if a habitat or species is very adversely affected by an external factor arising from human activities or natural events (killed/destroyed, 'high' intolerance) and is expected to recover over a very long period of time, i.e. >10 or up to 25 years ('low' recoverability) then it would be considered to be highly sensitive. Similarly, if a habitat or species is adversely affected by an external factor arising from human activities or natural events (damaged, 'intermediate' intolerance) but is expected to recover in a short period of time, i.e. within one year or up to five years ('very high' or 'high' recoverability) then it would be considered to be of low sensitivity.

Biological Traits Analysis -Community Analysis

The ecological traits expressed by macroinvertebrates have been used to explore the effects of impacts on the functioning of benthic, macroinvertebrate assemblages. Tillin et al. (2006) used biological trait analysis to explore the effects of fishing in North Sea soft sediment habitats. This study showed that community level changes in the functional trait expression of the community could be linked to fishing effects. Similarly Bremner et al. (2003) used trait analysis to compare fished and unfished areas in the

North Sea and found that increases in fishing effort changed the functional composition of the assemblage.

Table 37 Combining 'intolerance' and 'recoverability' assessments to determine 'sensitivity'

Intolerance	***Recoverability***						
	None	***Very low (>25 yr.)***	***Low (>10–25 yr.)***	***Moderate (>5 -10 yr.)***	***High (1 -5 yr.)***	***Very high (<1 yr.)***	***Immediate (< 1 week)***
High	Very high	Very high	High	Moderate	Moderate	Low	Very low
Intermediate	Very high	High	High	Moderate	Low	Low	Very Low
Low	High	*Moderate*	*Moderate*	Low	Low	Very Low	Not sensitive
Tolerant	Not sensitive	Not sensitive	Not sensitive	Not sensitive	Not sensitive	Not sensitive	Not sensitive
Tolerant*	Not sensitive*	Not sensitive*	Not sensitive*	Not sensitive*	Not sensitive*	Not sensitive*	Not sensitive*
Not relevant	*Not relevant*	*Not relevant*	*Not relevant*	*Not relevant*	*Not relevant*	*Not relevant*	*Not relevant*

Both studies used multivariable analyses to explore differences in trait composition (functional diversity) in the biological assemblage between areas subject to different levels of fishing intensity. As with the qualitative and empirical models described above these studies have provided useful information to inform sensitivity assessments (for example, they have shown that fishing leads to losses of attached epifauna, reduction in suspension feeders, loss of large and long-lived organisms).

The method developed by Marine Ecological Surveys Limited to predict the sensitivity of species based on traits was described in the species level section (Section 6.1). These species assessments were used as the basis of community level assessments of recoverability (Marine Ecological Surveys Ltd 2007).

Analysis of the proportion of biomass of each taxon from each site for which information had been collected showed how much of the measured biomass at each site possessed each trait. This allowed the ratio between the proportion of biomass that is represented by vulnerability and recoverability traits to be calculated as an index of sensitivity.

Sensitivity Matrices: The 'Beaumaris Approach':

CCW developed a methodology, referred to hereafter as the 'Beaumaris approach' for assessing the sensitivity of benthic habitats to fishing impacts based on a workshop held at Beaumaris, North Wales, in March 2006. Following a trial, a modified approach was used by the University of Liverpool (Hall et al. 2008), based on expert judgement, to determine habitat sensitivity to fishing impacts. The Beaumaris approach used a matrix approach where a single matrix, was used, with one axis composed of 14 types of fishing and aquaculture activity and the other of marine habitat types (found in Wales).

Sensitivity was assessed based on data available from published scientific literature and unpublished reports. The assessment is based on various factors:

- the likely degree of physical disturbance to seabed structures and sediment;

- the size of area damaged;
- the effects on sediment structure and faunal assemblages, reduction in species diversity;
- the effect on non-target fauna; and
- the loss of long-lived, slow growing species and biogenic habitat features.

The sensitivity assessment considers all aspects cumulatively.

Sensitivity was scored as low, medium and high or not applicable (gear type unlikely to occur in habitat type). The basis for the sensitivity score was provided alongside the matrices, to add robustness and transparency to the methodology. Hall et al. (2008) describe their approach as broadly consistent with MarLIN and SensMap but that the assessment was also based on the author's expert opinion.

The main steps of this method are:

i. to conduct a literature review to assess impacts, data extracted relating to the decline in abundance of a species from a disturbance event (also used information on decline in phyla, families and general levels). Abundance data were used to determine the intensity of impact. The size of area damaged formed the spatial extent axis of the matrix.

ii. the results of the literature review are interpreted by expert judgement to assess the sensitivity of each habitat type against an intensity scale for the impacting activity (fishing in that study). The output from this sensitivity assessment is a series of matrices of habitat type against impact and intensity (Hall et al. 2008).

This methodology was used by Tyler-Walters and Arnold (2008) to evaluate the sensitivity of intertidal habitats to foot and vehicular access for CCW. Sensitivities of 16 intertidal habitats were assessed (from the major intertidal habitat types identified by Hall et al. 2008). Seven types of access were considered e.g. foot, bicycles, motorcycles, all terrain vehicles, tractors etc. For each access type intensity was scaled according to the existing scale for 'hand gathering' in the Hall et al. (2008) sensitivity assessment (from heavy; access by >10 people per hectare per day, to single; access on a single occasion.).

The approach is based on expert judgement and informed by the scientific literature. These studies provided an assessment of the sensitivity of features to pressures that were used subsequently to develop sensitivity maps by CCW through a process of expert workshops.

6.2.4 Regional Sensitivity Assessments Incorporating Non-biological Features.

A number of sensitivity and vulnerability indices have been developed that incorporate biological and non-biological features, particularly in relation to oil spills. Two approaches are discussed briefly below and summarised in Table 38 (below).

Oil Spill Vulnerability Index

The oil spill vulnerability index of Gundlach and Hayes (1978) integrated geological vulnerability (based on deposition, penetration and persistence of spilled oil) and biological vulnerability (based on the species-specific sensitivity to spilled oil, length of exposure, rate of biological recovery and toxic properties of spilled oil). The resulting classification provides a qualitative ranking of different shore types where exposed rocky headlands were the least vulnerable. Mapping according to the Vulnerability Index mapping of shore types within an area indicates the type of treatment required for oil spills allowing the rapid deployment of protection where required, e.g. oil containment booms should be placed to protect sheltered tidal flats and salt marshes.

Table 38 Summary of regional assessments of sensitivity incorporating non-biological features

Metrics	Description	Examples
Oil Spill vulnerability index	Integrates geological vulnerability and biological vulnerability to spilled oil using a number of characters to define a 1-10 scale of sensitivity.	Gundlach and Hayes 1978; Davis et al. 1980
Environmental Sensitivity Index (ESI)	Assesses coastline sensitivity to oil spills, based on a shoreline classification (exposure, sediment/substrate features, biological productivity and sensitivity), biological resources (importance of shoreline to oil sensitive species) and human use resources. Information collated on biological resources is used to inform natural resource management.	Lindstedt-Siva et al. 1983, Michel and Dahlin 1993, Schiller et al. 2005

The resultant guidelines indicate that 'case reports' supporting the nomination of habitats and species should include information on the degree of threat, cross-referenced to the decline criterion and information on whether the threat is attributable to human activities (OSPAR 2003). Habitat descriptions should be cross-referenced to at least EUNIS level 3 but ideally 4 or 5.

Environmental Sensitivity Index (ESI)

The Environmental Sensitivity Index (ESI) was developed by the NOAA of the U.S. Department of Commerce. ESI atlases have been prepared most of the US coastline, including Alaska and the Great Lakes and are used to inform oil-spill contingency planning and response as well as providing information for coastline management.

ESI datasets are comprised of Shoreline-Classifications, ranked according to a scale relating to sensitivity, natural persistence of oil and ease of clean-up. The following factors determine the sensitivity ranking.

i. Relative exposure to wave and tidal energy

ii. Shoreline slope

iii. Substrate type

iv. Biological productivity and sensitivity.

Biological resources include oil-sensitive plants, animals, and habitats which are used by oil-sensitive species, including biogenic habitats which are oil-sensitive such as submerged aquatic vegetation and coral reefs. For these species, sensitivity assessments have regard to habitats where:

i. abundances are high;

ii. habitats are important for marine/aquatic specific at life stages spawning, nesting, resting, molting;

iii. early life stages or migration pathways occur in restricted areas;

iv. specific areas are sources of propagules;

v. species are threatened/endangered/rare;

vi. a significant percentage of threatened/endangered species exist there.

Human-Use Resources specific areas have added value because of their use, such as:

i. high use recreational and shoreline access locations (beaches, parks, boat ramps);

ii. management areas (i.e. refuges, preserves, sanctuaries, etc.);

iii. resource extraction locations (water intakes, aquaculture etc.);

iv. archaeological, historical and cultural resource sites.

6.3 Mapping Sensitivity and Vulnerability

The previous section has reviewed methods of assessing the sensitivity of species, habitats and broader regions to pressures that cause impacts. In Section 2, the difference between sensitivity and vulnerability was clarified. A species or a habitat may be sensitive to a particular pressure but if that pressure does not overlap with the occurrence of the species or habitat or does not take place in the region, then the ecosystem components are not vulnerable to that pressure. This means that sensitivity assessments alone are not informative for marine spatial planning.

In relation to marine spatial planning, sensitivity assessments should be undertaken alongside pressure mapping to evaluate the vulnerability of the selected ecosystem components. In recent projects, the spatial distribution of pressures has been assessed and related to sensitivity. Recent studies have been undertaken by the Centre for Environment, Fisheries and Aquaculture Science (Cefas) to assess the broad-scale distribution of pressures on the UK continental shelf. This includes work by Eastwood et al. (2007) using spatial data for the major human activities occurring in the England and Wales sector of UK waters in 2004, to provide an assessment of direct, physical pressure on the seabed from multiple human activities. Stelzenmuller et al. (2008) assessed the spatio–temporal distribution of fishing pressure on marine landscapes in UK waters (England and Wales) based on VMS data.

In this section, two approaches to delivering vulnerability assessments are described as outlined in Table 39.

Table 39 Examples of Vulnerability Assessments

Metrics	Description	Examples
Cumulative Effects Assessment (CEA)	CEA uses a matrix of activities (pressures) and associated possible environmental effects (impacts) on selected valued Ecosystem Components (species and habitats)	Oakwood Environmental Ltd (2002)
Robinson et al. (2008)	Pressure assessment methodology developed for UKMMAS/OSPAR. An overall assessment of vulnerability is derived using an assessment of exposure (overlap between pressure extent and component) and sensitivity (resistance and resilience).	Robinson et al. (2008)

Cumulative Effects Assessment

Oakwood Environmental Ltd (2002) undertook a study for CCW to provide a Cumulative Effects Assessment (CEA) of the offshore industries of Liverpool Bay. CEA uses a matrix of activities (pressures) and associated possible environmental effects (impacts). A requirement for assessing cumulative effects has been stipulated in EC Directive 97/11/EC, amending the EIA Directive 85/337/EEC in which the criteria for assessment include looking at 'the cumulation with other projects'. The 1992 Habitats Directive (EC Directive 92/43/EEC) also stipulates that an ''appropriate assessment'' for a plan or project investigates effects 'in combination with other plans or projects'.

The project used a geographical information system (GIS) package to store matrices relating to the effects assessment and the results displayed as a layer on a map of the study area.

Three Valued Ecosystem Components (VECs) were identified to provide a focus for assessment: the potential feeding habitat for the common scoter, the spawning habitat for plaice and sole, and the visual seascape value. The VECs were selected on the basis that they are importance receptors to the known activities and help guide the CEA process by identifying issues for the assessment.

The vulnerabilities of the VECs were calculated for each pressure, from all current and proposed activities in the Liverpool Bay study area. Vulnerability uses a matrix scoring vulnerability for habitats and species based on the English Nature methodology (used for assessing impacts on features of potential Special Areas of Conservation (pSAC) in Habitats & Conservation Act Regulation 33 documentation), where vulnerability is a function of exposure multiplied by sensitivity (V1). An additional step to the English Nature methodology was added by Oakwood Environmental Ltd. to take recoverability into consideration (V2).

Vulnerability is calculated using the following formulae:

$$\mathbf{V1 = E \times S} \qquad \text{(Equation 5)}$$

$$\mathbf{V2 = V1 \times R} \qquad \text{(Equation 6)}$$

Where:

V1 = vulnerability (not weighted against recoverability).

V2 = vulnerability (with recoverability rating).

E = Exposure to effects - assessed using ranking criteria based on scientific understanding, distance from activity and natural processes to create buffer zones.

S = Sensitivity - based on MarLIN sensitivity biotope scores.

R = Recoverability - based on MarLIN biotope scores

The output has provided a series of maps which identify areas of Liverpool Bay which are likely to be vulnerable to certain effects at different timescales, these being: current (2002); during wind farm construction (2003) and 2005 when operation of the wind farms could commence. It has not been possible to determine the significance of the suggested cumulative effects as this is beyond the scope of this report.

Robinson et al. (2008) Methodology

The pressure assessment methodology developed by Robinson et al. (2008) for UKMMAS/OSPAR uses expert judgement to produce an assessment of exposure (overlap between pressure extent and component) and sensitivity (resistance and resilience) to give an overall assessment of vulnerability. The degree to which a component responds to a pressure represents sensitivity and is a function of resistance and resilience. The outputs of this work are not currently available but the methodology can be generally outlined.

The methodology:

i. pressures acting in the region are described in terms of their spatial distribution, extent, frequency, seasonality and intensity.
ii. the spatial distribution, extent, frequency, seasonality and intensity of the ecosystem component being assessed, is described.
iii. exposure to each pressure is considered in terms of actual spatial overlap. If there is no overlap between the pressure and the component then no further evaluation is needed.
iv. where there is spatial overlap the component response to the pressure is evaluated in terms of resistance and resilience, to determine the final score for a pressure/component interaction (this step is outlined further below).

The degree of impact/threat (vulnerability) can be interpreted in terms of current status relative to a historic baseline or risk to the component based on future threats.

Degree of impact/threat = (Exposure Resistance Resilience) (Equation 7)

Resistance is categorised as high or low and is defined by an acceptable threshold limit for the component to the pressure, given the status of the component relative to a baseline and the degree of exposure to the pressure. Robinson et al. (2008) stated that thresholds should relate to the overall objective of maintaining good environmental status against a background of sustainable use.

Resistance thresholds to assess current status relative to former natural conditions were derived from elements of Favourable Conservation Criteria (FCC) and OSPAR Texel-Faial criteria.

- The geographic range of the habitat in the region being assessed should not have decreased by >10 per cent compared to former conditions.
- The total area of habitat lost should not exceed 15 per cent of the baseline area.

And/or

- No more than 25 per cent of the baseline area of habitat should be damaged as a result of a pressure (if there is no loss).

Resilience is defined by recovery time in terms of four ranks based on the range of recovery times seen in marine ecosystem components. Recovery is assessed to a point just beyond the resistance threshold level. For example, for habitats, this is the recovery time necessary to recover by 5 per cent of the existing area of extent, or 10 per cent in terms of area damaged if considering recovery over a relatively short period (i.e. the next 10-20 years) (Table 40).

Having completed Steps i - iv, a 2 x 4 matrix of resistance and resilience categories can then be used to select the degree of impact or future threat.

Table 40 Recovery Time Categories

Rank	Criteria
(1) No resilience	No recovery observed over any time scale following any change in the component.
(2) Low resilience	Recovery time is between 10 and 100 (+) years. This is observed for biological components with K-type life history strategies where loss of biomass or abundance is associated with very long recovery periods.
(3) Medium resilience	Recovery time is between two and 10 years. This is observed for biological components with life history strategies that are relatively productive; for example, many commercially exploited fish stocks.
(4) High resilience	Recovery time is less than two years. This is appropriate for biological components with r-type life history strategies where recruitment and turnover rates are high. For example, this is observed in some benthic invertebrate species with short generation times and high fecundity.

It is suggested that sensitivity assessments for each component should be performed twice – once based on an aggregate response and once on a worst-case. Most ecosystem components can be broken down into a number of more discrete sub-components (although this could go as far as a species by species assessment, if required). The aggregate assessment takes account of the sensitivity of a component based on the majority response of all sub-components, whilst the worst-case is based on the sub-component most sensitive and also exposed to the pressure.

The assessment is based on based on expert judgement and, to reflect this, confidence assessments are assigned to the evaluations. This is an expert-judgement based approach, relying on a semi-quantitative assessment of a number of aspects of the pressure/component relationship and based on an audit trail of decisions made at each step. The confidence assessment reflects state of knowledge available for each assessment.

High confidence should be given when data are available, particularly in the form of GIS outputs for the period being assessed, and/or a group of experts (> three) agree that they have high confidence in the assessment. Where detailed information is not available for the period being assessed, or is not available at all, and/or there is no agreement, or the number of experts involved is < four, the confidence assessment is low.

6.4 Strengths and Weaknesses of Approaches

The review of sensitivity approaches has considered a range of methodologies that have been developed for disparate purposes and which utilise a range of information to deliver a sensitivity assessment. The main objective of the review is to identify methodologies of assessing sensitivity/vulnerability that will underpin management of commercial fisheries, to protect sensitive habitats and deliver good ecological status.

We suggest that there are three main elements of a sensitivity assessment that should be considered in order to identify whether an approach is suitable or not. These are:

- fitness for purpose
- degree to which approach can be understood by, and justified to, stakeholders
- ease of implementation

We have considered each sensitivity assessment methodology with regard to these criteria and the strengths and weaknesses of each of the approaches are summarised in the project spreadsheet. The rationale for the information considered in this table, and its relevance to the commercial fisheries risk assessment, is discussed briefly below. The table headings and contents are described in Table 41.

Table 41 Structure of the strength and weakness spreadsheet

Element	Description	Table Heading
Fitness for purpose	Discriminate between fishing impacts on different habitat types.	Applicability to Fishing Impacts, Applicability to Habitats
	Discriminate between gear types and intensities	Gear Types
	Does the assessment use information on recovery and of which parameters/elements?	Recovery
	Does the assessment incorporate information on the resistance and if so, which parameters/elements?	Resilience
	Can the assessment be used to deliver a broad-scale assessment?	Spatial Scale
Communication to Stakeholders	Can the approach and outputs be readily communicated to non-specialists?	Ease of interpretation for non-specialists
Implementation	The level of information required, e.g. habitats, species,	Taxonomic requirements
	Type of information required	Information requirements
	Any specific issues relating to implementing the assessment.	Ease of Implementation

The strengths and weaknesses of the approaches described in this review are assessed with regard to the purposes of the commercial fisheries risk assessment for the WFD. The strengths and weaknesses are summarised below (Table 42) and are detailed in the project spreadsheet.

It is expected that sensitivity assessments will ultimately be combined with information on the distribution of habitats and fishing activities to identify vulnerability or risk. In order to be fit for purpose an approach should identify:

i. **Compatibility** between fishing activities and habitats, e.g. where fishing activities are sustainable or can be supported. Therefore, the assessment will allow managers to identify where fishing activities can occur to minimise impacts.

ii. The assessment should also indicate **incompatibility,** by identifying which habitats are most sensitive and to which gear types, so that areas where fishing types/intensities should be excluded or subject to management controls are identified.

In order to identify risk, it is essential that the approach should **discriminate** between the **sensitivities of different habitats to fishing impacts.** The discrimination may take the form of sensitivity categories, rank or scores for habitats (approaches may contain different steps and do more than one of these, e.g. score, then categorise habitats). It is also desirable that the assessment should be able to **discriminate** between **different gear types and intensities,** which should provide a more utilitarian approach to mapping sensitivity and vulnerability.

Although delivery of these objectives is the primary requirement of an approach to sensitivity assessment, there are other desirable characteristics.

Stakeholders should be able to understand an approach easily, as ‘buy-in’ is crucial to the success of management projects. Similarly, in order to gain support, the sensitivity assessment should be justifiable.

Table 42 Summary of the strengths and weaknesses of different sensitivity assessment methodologies

DESCRIPTION		SUMMARY	
Approach	**Description**	**Strengths**	**Weaknesses**
Species Level			
Biological Traits Assessment (Bremner et al. 2003, Tillin et al. 2008).	Sensitivity assessment of species based on life history traits. Habitat/biotope sensitivity assessed through multivariate analysis of traits expressed by constituent species.	Has been used to assess effects of perturbation, ecological function and describe sensitivity. Traits can be selected according to user requirements.	Based on literature review, comprehensive assessments of assemblages require a large amount of detailed species information. Assessments do not include information on habitat resistance and/or resilience as they are based solely on species.
Life Form Assessment (Holt et al. 1995).	Sensitivity of species or life forms assessed based on resistance and recoverability to impacting activities. Sensitivities are categorised and presented as a table, rather than combined to produce an index score.	The methodology provides a useful, transparent, audit to demonstrate how sensitivity has been assessed.	In this form, the approach would not deliver a usable assessment of sensitivity. Development of the methodology into a useable form is demonstrated by MacDonald et al. (1996), and McMath et al. (2000).

Table 42 (continued) Summary of the strengths and weaknesses of different sensitivity assessment methodologies.

DESCRIPTION		SUMMARY	
Approach	**Description**	**Strengths**	**Weaknesses**
Species Level			
MacDonald et al. Sensitivity Index (MacDonald et al. 1996).	Sensitivity of species to fishing by scoring fragility and recoverability. Scores are combined to provide a sensitivity index.	The resistance and resilience scores provide a useful audit trail to demonstrate how index score was derived.	Assessments do not include information on habitat resistance and/or resilience as they are based solely on species.
Habitat/Biotope Approaches			
Qualitative Models (Auster, 1998; Bax and Williams 2001).	Ranking of habitat sensitivity in relation to an impacting activity, Auster et al. (1998). Developed a hierarchical classification of habitat sensitivity (including biogenic habitats) to fishing gear based on evidence for impacts on complexity.	Simple, evidence based, qualitative assessment of habitat sensitivity, conceptually easy to understand and could be used to develop a qualitative broad-scale sensitivity assessment (based on habitat distribution).	No formal methodology for assessing sensitivity, due to qualitative nature management decisions based on assessment may be difficult to justify. In current iteration Auster et al. (1998) does not incorporate a measure of habitat recovery.
Empirical Models (Collie et al. 2000;Kaiser et al. 2006, Hiddink, et al. 2007)	Assessment of sensitivity to fishing impacts based on regression analyses of fishing gear types and habitat and using species and community attributes, e.g. abundance, species richness or attributes of ecosystem function as the response (sensitivity) factor.	Does not rely on expert judgement or species information. Provides quantitative information on impacts. Approach provides useful information to support sensitivity assessments.	Information requirements are high (requires experimental manipulations or numerous datasets) and currently preclude use of this approach. Further work required to develop and validate models,
Biological Traits Analysis (Marine Ecological Surveys Limited 2007)	Species assessments based on recoverability traits are used as the basis of community level assessments of recoverability	Increasing availability of species trait information through work undertaken by Marine Ecological Services, MarLIN and others, supports trait based approaches.	Based on recoverability rather than resistance, this is less appropriate for fishing where the degree to which a habitat/biotope can withstand (resist) impacts is of interest to managers. To specifically address fishing impacts, the approach would have to be tailored to assess sensitivity to gear types and different fishing intensities. Assessments do not include information on habitat resistance and/or resilience as they are based solely on species.

Table 42 (continued) Summary of the strengths and weaknesses of different sensitivity assessment methodologies.

DESCRIPTION		SUMMARY	
Approach	**Description**	**Strengths**	**Weaknesses**
Multivariable Indices			
SENSMAP Approach Based on MacDonald et al. (1996); further development by McMath et al. (2000)	The SENSMAP approach is based on the sensitivity index developed by MacDonald et al. (1996). It assesses sensitivity of biotopes/habitats to impacting activities by combining assessments of selected species sensitivity to produce an index score. Index scores are then ranked and categorised and assigned different colours to output colour coded maps of sensitivity.	Consistent, transparent approach already used to develop regional sensitivity maps (in the Irish Sea). Could be extended specifically to address fishing activities.	To specifically address fishing impacts, the approach would have to be tailored to assess sensitivity to gear types and different fishing intensities. Assessments do not include information on habitat resistance and/or resilience as they are based solely on species.
MarLIN Approach (Hiscock and Tyler-Walters 2006).	The MarLIN approach assesses the sensitivity of species and biotopes to 24 separate environmental factors (including physical, chemical and biological) by combining a measure of intolerance with a measure of recoverability. Sensitivity is ranked according to six categories	Sensitivity assessments for habitats/biotopes based on information of species intolerance and recoverability. Well-developed and widely supported methodology.	To specifically address fishing impacts, the approach would have to be tailored to assess sensitivity to gear types and different fishing intensities. Assessments do not include information on habitat resistance and/or resilience as they are based primarily on species.
Beaumaris Approach (Hall et al. 2008, Tyler-Walters and Arnold, 2008).	Assesses habitat sensitivity to a pressure (fishing), at different intensities, to derive a sensitivity score (high, medium, low). Uses a matrix containing three factors; fishing metier, intensity of fishing, and habitat sensitivity.	Approach was developed specifically to deliver sensitivity assessments for fishing impacts. The approach is defendable as decision making is transparent through the use of an audit trail. Approach includes information on habitat characteristics and resistance to fishing disturbance.	Assessments do not include information on habitat resilience as they are based solely on species. Reliance on expert judgement may be considered a weakness but in other assessments, use of indices may provide spurious confidence and these assessments may also be supported by expert judgement.

Table 42 (continued) Summary of the strengths and weaknesses of different sensitivity assessment methodologies.

DESCRIPTION		SUMMARY	
Approach	**Description**	**Strengths**	**Weaknesses**
Vulnerability Assessments			
Shore-Type Assessments (Gundlach and Hayes, Davis et al. 1980)	Integrates geological and biological sensitivity to impacting activity using a number of characters to define a 1-10 scale of sensitivity. The resulting classification provides a qualitative ranking of different shore types.	Information and spatial mapping of physical habitat attributes, more widely available than information on biological attributes. An approach that includes habitat characteristics may therefore be useful for spatial mapping	The approach would have to be developed (and tested) to address fishing impacts.
Environmental Sensitivity Index (ESI) (Michel and Dahlin 1993, Schiller et al. 2005).	Assesses coastline sensitivity to oil spills, based on a shoreline classification (exposure, sediment/substrate features, biological productivity and sensitivity), biological resources (importance of shoreline to oil sensitive species) and human use resources. Information collated on biological resources is used to inform natural resource management.	Information and spatial mapping of physical habitat attributes, more widely available than information on biological attributes. An approach that includes habitat characteristics may therefore be useful for spatial mapping	The approach would have to be developed (and tested) to address fishing impacts.
Vulnerability			
HBDSEG Approach (Robinson et al. 2008)	Pressure assessment methodology developed for UKMMAS/OSPAR. An overall assessment of vulnerability is derived using an assessment of exposure (overlap between pressure extent and component) and sensitivity (resistance and resilience).	Provides audit trail and confidence assessment for conclusions. Baselines for assessment would require further development and would be informed by management purposes.	The approach would have to be developed (and tested) to address fishing impacts. Based on expert judgement so success would be constrained by availability of knowledge

Preferentially an approach should be demonstrably evidence based so that there is the highest possible confidence in the assessment. Therefore, the degree to which an assessment is based on expert judgement should be indicated. As, inevitably, there will be information gaps the methodology should be transparent (preferably with audit trails of decision making, so that assessments can be updated where new evidence is gathered) be repeated and be justified. Allied to this, if confidence assessments can be provided these will indicate where there is uncertainty in the assessment as well as areas where there is high confidence in the results.

Consideration should also be given to the ease of implementation / operation of sensitivity approaches.

7 Information Gaps

The last decade has seen a large increase in the amount of data concerning the effects of fishing activities on marine habitats in both intertidal and the subtidal. Nevertheless, there remain numerous gaps in our information and understanding that any approach to sensitivity assessment needs to address and manage.

Distribution of Habitats and Biotopes

Full coverage of the UK seabed is currently only available at EUNIS level 3. Much of this coverage is based on modelled habitat distribution. It may not represent biological communities. Inshore areas relevant to WFD are more likely to have additional information available than offshore areas. Intertidal areas are much better covered to EUNIS level 4 and 5 due to the MNCR survey of the 1990s and subsequent surveys by statutory agencies.

Distribution of Inshore Fishing Fleet

To produce vulnerability maps data on type and distribution of fishing effort is required. The Sea Fisheries Committees (SFC) regulate these activities in inshore waters and will hold some information, although the level of detail varies between SFC regions. The large amount of information available from VMS, and held by Cefas was, until recently, only collected for vessels over 15 m in length, and misses much of the inshore fleet. Cefas have however been doing further work to map inshore fin fisheries. ABPmer have also produced a data layer on inshore fisheries value (Dunstone 2009) so that the locations of shell-fishing are known. Data on recreational angling and recreational activities affecting the intertidal are also missing.

Information on Sensitivity to Fishing Impacts

The habitat reviews developed for this report identified the following gaps in information.

Habitat/Location

- Few studies have examined the impacts of fishing activities on shallow mud communities.
- Few studies have examined ‘Vertical surfaces’ or ‘Chalk reefs’.
- Very few studies have used experimental sites or data from deep water locations (although this is less relevant for inshore fisheries).
- correct diagnosis of the source of seagrass bed alteration is important for the implementation of management measures (Ardizzone et al. 2000).
- A detailed review of the extent, distribution and status of European maerl beds is still needed (Hall-Spencer and Moore 2000a).
- Invertebrate diversity needs to be studied on much larger scales, equivalent to those that have been used for fish, and to separate fishing and biogeographical effects, together with a need for more accurate information

of fishing effort distribution through satellite tracking (Greenstreet and Rogers 2000).

Ecosystem structure and functioning

Information on the basic biology of many dominant or abundant benthic organisms remains limited. Biological traits of relevance to recoverability (e.g. life history characteristics) often have to be inferred from similar species (congeners, or familials). Identifying keystone and characterising species, the factors affecting interconnectivity and larval supply between areas, and the importance of meta-populations are unclear, and likely to vary around the coast of the UK. Further basic research into the autoecology of many species is still required, even species of conservation concern. Open Access resources such as the Biological Traits Information Catalogue (BIOTIC[3]) that collate and disseminate traits information could provide a platform to share such information throughout the marine research community.

Parameters for Measuring Response to Fishing

It is clear from the vast number of parameters used in the studies reviewed (Table 27) that there is no single agreed descriptor to effectively measure the effects of fishing on species and habitats. Furthermore, very few indicators are exclusive to fishing impacts. For all indicators, much theoretical and analytical work is required to design reference points (Rochet and Trenkel, 2003). An alternative approach to using single indicators is to examine multiple indicators to accumulate evidence (Rice, 2000).

There is still much to be done in developing indicators of fishing impacts on fish communities. Indicators need to be improved and different methods properly investigated. Providing a theoretical framework for linking fishing to community metrics is also essential to provide unambiguous predictions of how particular indicators should change with fishing (Rice, 2000). There is an urgent need to incorporate science in the assessment process to avoid the prevalence of irrational and prejudiced views (Rochet and Trenkel, 2003).

Fishing Gear and Intensity

Extensive research has focussed on the effects of towed fishing gears on benthic communities but there is less advice available on tolerable levels of fishing (Sewell et al. 2007).

Many techniques can be used to reduce by-catches and the impacts of fishing gears operating on the seabed. However, it is difficult to generalise solutions and make them applicable to all metiers in the fishery. As gears and local circumstances differ greatly, solutions should be aimed at specific sectors in the fishing industry (Van Marlen 2000).

Most studies looked at for the basis of this report focussed on mobile gears (e.g., trawling, dredging). Studies looking at static gear types such as potting and creeling (e.g., Eno et al. 2001) were less common. Further site-specific studies on the effects of potting and other static gears would be required to determine the optimum fishing levels that would satisfy fisheries and nature conservation interests (Eno et al. 2001).

Identification of impacts by trawls is the first step in a lengthy management process that must consider not only the ecology of the affected taxa, but various socioeconomic

[3] BIOTIC – http://www.marlin.ac.uk/biotic

impacts as well. In the United States, resource managers are required to minimize adverse impacts of fishing (and other anthropogenic disturbances) on essential fish habitat, to the maximum extent practical (McConnaughey et al. 2000).

Most studies looking at the effects of gear type are only able to sample a proportion of the benthic community, i.e., that retained as by-catch in commercial fishing gear. In order to draw conclusions about the whole community structure a more comprehensive and targeted approach to sampling would be required, e.g., using a combination of grab sampling and fine-mesh beam trawling (Veale e*t al*., 2000).

Few published studies (e.g., Collie et al. 2000; Jennings et al. 2001) have investigated the effects of towed fishing gears on real fishing grounds, but no such studies appear available for hydraulic or scallop dredging in UK waters (Sewell et al. 2007). Such comparative impact studies have only become possible due to the release of over-flight or satellite vessel monitoring data (VMS) (e.g. Eastwood et al. 2007). VMS data only accounts for relatively large vessels however, and so for studies to include inshore hydraulic and scallop dredging, higher resolution data including smaller fishing vessels will be essential (Sewell et al. 2007). Such data would allow the comparison of similar areas that differ in their fishing intensities. Sewell et al. (2007) advised that comparative fishing effort studies should be supported in the future and that the fishing industry should encourage the collection and release of high resolution effort data to science. Progress can then be made towards establishing tolerable and sustainable levels of fishing.

Spatial and Temporal Studies

Many studies looked at in this report have used a small-scale short-term approach, where a section of the seafloor is trawled by a single pass or multiple passes, and damage is compared before and after or an experimental site with a control site. Results from such studies can provide an indication of the severity of initial impacts and relative rates of recovery, but are unsuitable for use as a precise management tool (Sewell et al. 2007).

A major limiting factor in many comparative fishing studies is the lack of locations where no fishing activity has previously occurred, i.e., lack of undisturbed pristine habitats. Low levels of fishing effort may have significant effects on the diversity and structure of fish communities. The greatest effects are most commonly observed when a previously unfished area is fished for the first time (Jennings and Kaiser, 1998). However, when an ecosystem is already in a fished state, diversity, structure and fish production tend to remain relatively stable across a wide range of fishing intensities, despite fluctuations in component species, which are driven by environmental changes (recruitment) and targeted overfishing (Jennings and Kaiser, 1998). Consequently, studies of fishing effects which begin in fished systems, and seek to detect changes with increasing fishing effort, will often suggest that fishing has limited effects on community structure and that the system is remarkably resilient. Only when fishing effort is so high that numerous species are depleted, or when fishers resort to habitat destructive fishing techniques, will further changes in community structure become apparent (Jennings and Kaiser, 1998).

The scale at which many experiments have been conducted is often not representative of the scale and intensity of real fishing grounds (Sewell et al. 2007). Recovery rates on real fishing grounds will be much longer and will depend heavily on larval supply rather than on migration for recovery. Due to the chronic nature of most fisheries, benthic communities may remain within an altered and mostly less productive state. The only way to test fully the effects of fishing on benthic communities is through large-scale experiments in areas closed to fishing (Ball et al. 2000).

A critical evaluation of the role of fishing in bringing about long-term changes in marine ecosystems is difficult to achieve (Frid and Clark 2000). Long term effects can only be determined with certainty with a long-term study in which changes to regions from which the impact is removed are monitored for many years with proper controls. Effective studies could last as long as the longest-lived component species, but frequently this is unknown (Currie and Parry 1996).

Mechanisms should be in place to provide for a more local management of fisheries, including those outside territorial waters. Fishing activity should probably be zoned further in both a time and spatial sense; these zones would include some no-take zones, where nature could develop with minimal interference from humans (Tasker et al. 2000).

8 Conclusions

Any assessment process or procedure, from EIA, Strategic Environmental Assessment (SEA), Risk Assessment (RA) or Vulnerability and/or Sensitivity Assessment, is an exercise in information collation, management and evaluation designed to enable or support decision making. All assessment procedures aim to use the best available scientific evidence but inherent in the procedure is the fact that we do not possess a complete understanding of the system we are trying to assess.

Therefore, sensitivity assessment in the marine environment has often been seen a 'holy grail', designed to synthesise all available information, in a transparent and systematic manner in order to allow users to decide on conservation or management priorities.

Numerous procedures have been developed over the past decades to address sensitivity of marine habitats. Yet all have been limited by the evidence base (on marine impacts, fishing intensity and community response), and design constraints so that they are not universally applicable.

As a result, no one method has gained favour with the marine environmental management community as a whole, and even less within the marine science community. The marine environmental management community has long sought a methodology that can be applied to all situations, at all scales, and while the assessment procedures, as shown in Section 6, are all designed to answer questions at prescribed scales and for specified activities. The scientific community, however, has tended to avoid 'expert judgement' and 'ranking' exercises in favour of more empirical assessment procedures that give less scope for bias.

The principle aim of this review was to examine the feasibility of a procedure to address the vulnerability of habitats to commercial fishing activities as part of the WFD commercial risk assessment. While there is an evidence base for fishing impacts on habitats (reviewed in Section 3 as resistance and recovery of biogenic features and geomorphological sedimentary features), in general much more interest has been focussed on impacts on the biological components of habitats. In order to be comprehensive we have therefore considered both species and community level sensitivity parameters and sensitivity assessments based on species and the biological assemblage. Species create much of the physical habitat structure on the seabed, e.g. pits, burrows, biogenic reefs, and are impacted by fishing impacts on habitat complexity. Therefore, biological sensitivity is an important component of any assessment of sensitivity to morphological impacts. However, any examination of morphological impact needs to address sensitivity to physical habitat modification (e.g. substratum change or modification) and the physical aspects of its natural restoration (e.g. sediment supply, water flow).

The above review leads to the following general conclusions.

i. Clear definitions of terms are vital for any assessment procedure, and terms such as resistance, intolerance, resilience recoverability, sensitivity and vulnerability need to be carefully defined and explained.

ii. Habitat groups (derived from the UK marine habitat classification) provide discernable units for assessment that are relatively easy to explain to stakeholders from multiple marine sectors, including the public.

iii. Biogenic habitat and those habitats dominated by long-lived, slow growing species are amongst the most sensitive to damage by fishing activities.

Due to their prolonged recovery period, maerl beds may be best viewed as non-renewable resources, lost forever if removed or destroyed.

iv. Soft sediment habitats vary in sensitivity depending on the mobility or cohesiveness of the sediment as well as the nature of the communities they support. The rate at which the physical habitat 'recovers' from damage is an important component of the rate at which the habitat as a whole is able to recover.

v. The impacts of fishing activities of chalk reef habitats are the least well studied of all the habitats examined.

vi. A large number of parameters (130) have been used in the 70 past studies examined to assess the sensitivity of habitats. These range from physical, chemical and biological parameters and include estimators of community structure and function.

vii. Nevertheless, no single descriptor or parameter can effectively or reliably explain the impact of fishing on community structure and habitat response. A number of parameters are required to describe the nature of the activity, the nature of the impact or response, the potential rate of recovery and overall sensitivity.

viii. The most used parameters include a suite of biological traits that describe a species or habitat (especially morphology and environmental position), their life history, the physical nature of the habitat itself (especially for soft sediments), and their contribution to ecosystem function (e.g. biogenic habitats) and function (e.g. biomass and productivity).

ix. However, biological traits alone cannot necessarily capture all aspects of the sensitivity of marine habitats, due to lack of data and understanding, and there is an important role for expert judgement in the assessment procedure.

x. Recent meta-studies and empirical studies on the effects of fishing on marine habitats (primarily soft sediments) has significantly improved our understanding of the relationship between fishing intensity, gear type, substratum type and impact and recovery.

xi. Studies of the effects of different fishing intensities are underpinned by VMS data on the movement of fishing vessels.

xii. The setting of clear, well defined, thresholds is a vital part of the assessment procedure. Thresholds include definitions of fishing intensities and gear types but also include thresholds of damage (acceptable vs. unacceptable), scales of resistance, resilience, sensitivity and vulnerability.

xiii. The sensitivity assessment methodologies reviewed were all developed for specific purposes, to answer specific management questions. Therefore, they are not completely applicable outside their original design parameters.

xiv. Sensitivity assessment is designed to manage uncertainties and information gaps. Although our understanding of the effects of fishing has grown considerably over the last twenty years, information gaps remain.

xv. Nevertheless, the existing sensitivity assessment methodologies provide a wide range of tools that could be applied to vulnerability assessment. Therefore, the development of an approach to the assessment of the vulnerability of habitats to commercial fishing activities is feasible.

9 Recommendations

It is clear that many sensitivity and vulnerability assessment procedures have been developed in the marine environment. No one method can be applied directly to an assessment of the vulnerability of marine habitats to commercial fishing activities at the scale examined (the habitat group). The Beaumaris approach (Hall et al. 2008) is the closest in focus (i.e. developed solely to address fishing impacts) but does not address 'resistance' (recovery), habitat recovery, or vulnerability. Nevertheless, much of the expertise and techniques developed in the last ten years can contribute to the development of a vulnerability assessment procedure.

Recommendation 1 – the development of a risk assessment for commercial fisheries activities within WFD requires a targeted approach that builds on existing expertise.

Recommendation 2 – any sensitivity assessment procedure needs a clear definition of the management questions it is designed to answer, and decisions it is designed to support, the scale at which it is to be applied and hence its limitations outside that remit.

With this in mind, it is recommended that while acknowledging the risk assessment has distinct management aims, consideration should be given to engagement with other forums where sensitivity assessment frameworks are being developed for a range of issues. For example, the conservation agencies and others are working towards Marine Protected Area planning as part of the provisions of the Marine Bill. The results of 'Charting Progress 2', the 2010 review of the status of the UK's marine environment, will also create impetus for the development of frameworks for assessing sensitivity across a range of issues.

Recommendation 3 - engagement with other agencies and their approaches is a potentially cost-effective way of drawing on a range of expertise, producing a widely supported methodology while reducing replication of effort.

The following recommendations relate more specifically to methodological requirements.

Clear definitions of terms, and activities are also vital to the success of the approach and, especially, how it communicates with stakeholders. The terms 'resistance' and 'resilience' (Hollings 1978) are relatively robust terms with clear definitions. Intolerance, recoverability, sensitivity and vulnerability also have been carefully and exactly defined many times (Hiscock 1999; Laffoley et al. 2000; Tyler-Walters et al. 2005). But the latter definitions are open to perception, as many consider 'sensitivity' to be equivalent to 'intolerance' and most stakeholders are likely to have a preconception of what they mean by sensitivity. Similarly, 'vulnerability' is open to mis-interpretation and many users and stakeholders do not distinguish between 'vulnerable' and 'sensitive'. Therefore, the term 'risk' may be more appropriate, and is widely used. Both 'vulnerability' and 'risk' incorporate a component of 'hazard' (i.e. likelihood of impact occurring) and 'consequence' (i.e. likelihood of damage or sensitivity).

Recommendation 4 – the terms 'resistance' and 'resilience' should be used in preference to 'intolerance', 'recoverability', and 'risk' in preference to 'vulnerability'.

In addition, the approach designed must decide on the amount of expert judgement vs. empirical analysis included. Empirical techniques have vastly improved but still suffer from data gaps, and modelled approaches (e.g. Hiddink et al. 2007) are very limited in their scope (i.e. the scale to which they can be applied and number of parameters they include). But empirical approaches have the advantage of being more objective and hence potentially more defensible in law (if that is a requirement).

However, expert judgement has the potential to collate more data and can include more parameters, while also engaging with stakeholders and capturing local knowledge that an empirical methodology could miss. Expert judgement has the potential advantage of compensating for data gaps by capturing knowledge that is not in the public domain or in a digital format. Nevertheless, disagreement between experts can undermine the approach. Audit trails of decision making and the provision of confidence assessments can improve transparency and allow assessments to be repeated and updated based on new information.

Overall, the authors would suggest a mixed approach, in which a systematic approach is developed to assess habitat 'sensitivity' and 'risk', the approach is used to derive 'risk assessments' and then validated by peer review by a panel of experts.

Recommendation 5 – the approach developed should include a systematic approach to assess 'sensitivity' and 'risk' followed by expert validation.

The above discourse suggests two ways in which to take forward the development of a risk assessment methodology for the effects of commercial fishing activities on marine habitats under WFD. Both approaches assume the development of an ecological risk assessment method.

i. Redevelopment of the approaches of MarLIN, and Hall et al. (2008) to include fishing intensity data, to derive 'risk'.

ii. Developments of a more empirical approach based on biological, habitat and community traits, in a systematic manner, and fishing intensity data to derive risk, using peer review and stakeholder review.

The first approach would be the simplest to achieve but would be primarily expert judgement driven. It would involve the use of relevant expertise to review the evidence base and rank habitat groups by their sensitivity, exposure to fishing activities and hence risk, followed by expert working groups (on a habitat type basis) to review and amend the assessments made. This approach would be the quickest to implement, although it should be noted that the Beaumaris approach (Hall et al. 2008) has taken several years to develop and implement.

The second approach would require further research but has the potential to be more objective, and to answer the questions specifically required under WFD.

Recommendation 6 – any systematic risk assessment approach should include the following elements:

i. Clear definition of aims, limitations, and terms (as above)

ii. Clear definition of fishing activities – as defined by CCW (Hall et al. 2008)

iii. Clear definition of acceptable and unacceptable levels of habitat damage/loss against which to assess sensitivity, e.g. either based on damage to the habitat (e.g. MarLIN, Hiscock and Tyler-Walters 2006) or against WFD criteria (i.e. ecological status).

iv. Information on fishing intensity for the inshore and local fleets, including small boats and recreational activities.

v. Development of a suite of parameters to describe those biological traits of relevant species, physical traits of habitats, and community traits that affect a habitats likelihood of damage from fishing activities at a variety of intensities.

vi. Development of a suite of parameters to describe those biological traits of relevant species, physical traits of habitats, and community traits that affect a habitats rate of recovery from disturbance.

vii. Systematic use of these parameters in a conceptual model, regression models or multivariate (non-parametric) analyses to develop an overall estimator of sensitivity.

viii. Mapping and systematic characterisation of habitat groups (based on available habitat maps).

ix. Mapping of fishing intensity against sensitivity to derive exposure and risk.

x. Expert judgment at each stage to manage data gaps.

xi. Final peer review of resultant habitat risk maps.

A risk based approach would entail methodologies to derive probabilities of damage (resistance) and recovery (resilience), and hence sensitivities. However, the second approach would necessitate a longer development time.

Overall, there is enough data, evidence and expertise to develop a systematic approach to the assessment of the vulnerability (and risk) of marine habitats to commercial fishing activities. But, before a methodology can be developed, clear decisions need to be made about the management and conservation questions the approach needs to answer, and the user group (or stakeholder group) that will need to understand and implement management.

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List of abbreviations

ABPmer	Associated British Ports Marine Environmental Research Ltd
ANOVA	Analysis of Variance
CCW	Countryside Council for Wales
Cefas	Centre for Environmental, Fisheries and Aquaculture Science
CEA	Cumulative Effects Assessment
Defra	Department for the Environment, Food and Rural Affairs
EA	Environmental Assessment
EC	European Commission
EIA	Environmental Impact Assessment
EUNIS	European Union Nature Information System
GAM	Generalised Additive Models
GIS	Geographical Information System
GLM	General Linear Model
HBDSEG	Healthy and Biologically Diverse Seas Evidence Group
ICES	International Council for the Exploration of the Sea
JNCC	Joint Nature Conservation Committee
MarLIN	Marine Life Information Network
MNCR	Marine Nature Conservation Review
MPA	Marine Protected Area
OSPAR	Oslo and Paris Commission
RA	Risk Assessment
SAC	Special Area of Conservation
SEA	Strategic Environmental Assessment
SENSMAP	Sensitivity mapping of inshore marine biotopes in the southern Irish Sea (SensMap)
SFC	Sea Fisheries Committee
UKBAP	UK Biodiversity Action Plan
UKMMAS	UK Marine Monitoring Assessment Strategy
VEC	Valued Ecosystem Component
WFD	Water Framework Directive

Appendix 1

Relevant biotopes and their MarLIN sensitivities. UK biotope classification codes (1997 and 2004 versions) and EUNIS are given for comparison.

Habitat group	EUNIS code	2004 code	1997 code	Habitat name	Intolerance	Recovery	Sensitivity	Confidence
Biogenic reef								
Sabellarid worm reefs								
	A2.71	LS.LBR.Sab		Littoral *Sabellaria* beds				
	A2.711	LS.LBR.Sab.Salv	MLR.Salv	*Sabellaria alveolata* reefs on sand-abraded eulittoral rock	Intermediate	High	Low	Moderate
	A3.215	IR.MIR.KR.Lhyp.Sab	MIR.SabKR	*Sabellaria spinulosa* with kelp and red seaweeds on sand-influenced infralittoral rock	Intermediate	High	Low	Moderate
	A4.221	CR.MCR.CSab.Sspi	MCR.Sspi	*Sabellaria spinulosa* crusts on silty turbid circalittoral rock	Intermediate	High	Low	Low
Maerls								
	A5.511	SS.SMp.Mrl.Pcal	IGS.Phy.HEc	*Phymatolithon calcareum* maerl beds with hydroids and echinoderms in deeper infralittoral clean gravel or coarse sand	High	Very low	Very High	Moderate
	A5.512	SS.SMp.Mrl.Lgla	IGS.Lgla	*Lithothamnion glaciale* maerl beds in tide-swept variable salinity infralittoral gravel	High	Very low	Very High	High
Bivalve beds								
	A1.111	LR.HLR.MusB.MytB	ELR.MytB	*Mytilus edulis* and barnacles on very exposed eulittoral rock	Intermediate	High	Low	Moderate
	A1.221	LR.MLR.MusF.MytFves	MLR.MytFves	*Mytilus edulis* and *Fucus vesiculosus* on moderately exposed mid eulittoral rock	Intermediate	High	Low	Moderate
	A1.223	LR.MLR.MusF.MytPid	MLR.MytPid	*Mytilus edulis* and piddocks on eulittoral firm clay	Intermediate	Moderate	Moderate	Moderate
	A3.361	IR.LIR.IFaVS.MytRS	SIR.MytT	*Mytilus edulis* beds on reduced salinity tide-swept infralittoral rock	Intermediate	High	Low	Moderate

Habitat group	EUNIS code	2004 code	1997 code	Habitat name	Intolerance	Recovery	Sensitivity	Confidence
	A4.241	CR.MCR.CMus.CMyt	MCR.MytHAs	*Mytilus edulis* beds with hydroids and ascidians on tide-swept moderately exposed circalittoral rock	Intermediate	High	Low	Low
	A5.625	SS.SBR.SMus.MytSS	IMX.MytV	*Mytilus edulis* beds on variable salinity infralittoral mixed sediment	Intermediate	High	Low	Moderate
	A5.435	SS.SMx.IMx.Ost	IMX.Ost	*Ostrea edulis* beds on shallow sublittoral muddy sediment	Intermediate	Moderate	Moderate	Low
	A5.621	SS.SBR.SMus.ModT	MCR.ModT	*Modiolus modiolus* beds with hydroids and red seaweeds on tide-swept circalittoral mixed substrata	High	Low	High	Moderate
Eelgrass beds								
	A2.6111	LS.LMp.LSgr.Znol	LMS.Znol	*Zostera noltii beds in upper to mid shore muddy sand*	Intermediate	Moderate	Moderate	Low
	A5.5331	SS.SMp.SSgr.Zmar	IMS.Zmar	*Zostera marina*/*angustifolia* beds in lower shore or infralittoral clean or muddy sand	Intermediate	Moderate	Moderate	Low
Faunal turfs								
	none	none	CR.C.FaV.Bug	*Bugula* spp. and other bryozoans on vertical moderately exposed circalittoral rock	Intermediate	High	Low	Moderate
	A3.117	IR.HIR.KFaR.LhypRVt	IR.AlcByH	*Alcyonium digitatum* with a bryozoan, hydroid and ascidian turf on moderately exposed vertical infralittoral rock	High	High	Moderate	High
	A3.117	IR.HIR.KFaR.LhypRVt	EIR.SCAn	Sponge crusts and anemones on wave-surged vertical infralittoral rock	High	High	Moderate	High
	A4.121	CR.HCR.DpSp.PhaAxi	MCR.ErSEun	Erect sponges, *Eunicella verrucosa* and *Pentapora fascialis* on slightly tide-swept moderately exposed circalittoral rock.	High	Very low	Very High	Moderate
	A4.134	CR.HCR.XFa.FluCoAs	MCR.Flu	*Flustra foliacea* and other hydroid/bryozoan turf species on slightly scoured circalittoral rock or mixed substrata	Intermediate	High	Low	Moderate
	A4.1341	CR.HCR.XFa.FluCoAs.Paur	MCR.MolPol	*Molgula manhattensis* and *Polycarpa* spp. with erect sponges on tide-swept moderately exposed circalittoral rock	Intermediate	Moderate	Moderate	Moderate

Habitat group	EUNIS code	2004 code	1997 code	Habitat name	Intolerance	Recovery	Sensitivity	Confidence
	A4.2145	CR.MCR.EcCr.FaAlCr.Pom	MCR.FaAlC	Faunal and algal crusts, *Echinus esculentus,* sparse *Alcyonium digitatum* and grazing-tolerant fauna on moderately exposed circalittoral rock	High	Very high	Low	Low
Macroalgal dominated hard substrata								
	A1.123	LR.HLR.FR.Him	ELR.Him	*Himanthalia elongata* and red seaweeds on exposed lower eulittoral rock	Low	High	Low	Moderate
	A3.126	IR.HIR.KSed.XKHal	MIR.HalXK	*Halidrys siliquosa* and mixed kelps on tide-swept infralittoral rock with coarse sediment.	Intermediate	High	Low	Low
	A3.111	IR.HIR.KFaR.Ala	EIR.Ala	*Alaria esculenta* on exposed sublittoral fringe bedrock	Low	High	Low	Low
	A3.113	IR.HIR.KFaR.LhypFa	EIR.LhypFa	*Laminaria hyperborea* forest with a faunal cushion (sponges and polyclinids) and foliose red seaweeds on very exposed upper infralittoral rock	Intermediate	Moderate	Moderate	Moderate
	A3.115	IR.HIR.KFaR.LhypR	EIR.LhypR	*Laminaria hyperborea* with dense foliose red seaweeds on exposed infralittoral rock.	Intermediate	Moderate	Moderate	Moderate
	A3.122	IR.HIR.KSed.LsacSac	EIR.LsacSac	*Laminaria saccharina* and/or *Saccorhiza polyschides* on exposed infralittoral rock	Intermediate	Very high	Low	Moderate
	A3.123	IR.HIR.KSed.LsacChoR	MIR.LsacChoR	*Laminaria saccharina*, *Chorda filum* and dense red seaweeds on shallow unstable infralittoral boulders or cobbles	Intermediate	High	Low	Moderate
Mixed sediments								
	A5.131	SS.SCS.CCS.PomB	ECR.PomByC	*Pomatoceros triqueter, Balanus crenatus* and bryozoan crusts on mobile circalittoral cobbles and pebbles	Tolerant	NR	Not sensitive	High
	A5.135	SS.SCS.CCS.Nmix	CGS.Ven	Venerid bivalves in circalittoral coarse sand or gravel	Intermediate	High	Low	Moderate
	A5.122	SS.SCS.ICS.HchrEdw	IGS.HalEdw	*Halcampa chrysanthellum* and *Edwardsia timida* on sublittoral clean stone gravel	High	High	Moderate	Moderate
	A5.261	SS.SSa.CMuSa.AalbNuc	CMS.AbrNucCor	*Abra alba*, *Nucula nitida* and *Corbula gibba* in circalittoral muddy sand or slightly mixed sediment	Intermediate	High	Low	Moderate
Mud sediments								
	A2.312	LS.LMu.MEst.HedMac	LMU.HedMa	*Hediste diversicolor* and *Macoma balthica* in sandy	Intermediate	High	Low	Low

Habitat group	EUNIS code	2004 code	1997 code	Habitat name	Intolerance	Recovery	Sensitivity	Confidence
			c	mud shores				
	A5.241	SS.SSa.IMuSa.EcorEns	IMS.EcorEns	*Echinocardium cordatum* and *Ensis* spp. in lower shore or shallow sublittoral muddy fine sand.	High	Moderate	Moderate	Moderate
	A5.322	SS.SMu.SMuVS.AphTubi	IMU.AphTub	*Aphelochaeta marioni* and *Tubificoides* spp. in variable salinity infralittoral mud	Intermediate	Very high	Low	Low
	A5.342	SS.SMu.IFiMu.Are	IMU.AreSyn	*Arenicola marina* and synaptid holothurians in extremely shallow soft mud.	Intermediate	High	Low	Low
	A5.343	SS.SMu.IFiMu.PhiVir	IMU.PhiVir	*Philine aperta* and *Virgularia mirabilis* in soft stable infralittoral mud	Intermediate	Moderate	Moderate	Low
	A5.354	SS.SMu.CSaMu.VirOphPmax	CMS.VirOph	*Virgularia mirabilis* and *Ophiura* spp. on circalittoral sandy or shelly mud	Low	Very high	Very Low	Moderate
	A5.361	SS.SMu.CFiMu.SpnMeg	CMU.SpMeg	Sea pens and burrowing megafauna in circalittoral soft mud	Intermediate	High	Low	Moderate
	A5.363	SS.SMu.CFiMu.BlyrAchi	CMU.BriAchi	*Brissopsis lyrifera* and *Amphiura chiajei* in circalittoral mud	Intermediate	High	Low	High
Saltmarsh								
	C3.44	none	LMU.Sm	*Pioneer saltmarsh*	Intermediate	High	Low	Very low
	C3.44	none	LMU.Sm.SM13	*Puccinellia maritima* salt marsh community	Intermediate	High	Low	Low
Sand sediments								
	A5.123	SS.SCS.ICS.MoeVen	IGS.FabMag	*Fabulina fabula* and *Magelona mirabilis* with venerid bivalves in infralittoral compacted fine sand	Intermediate	High	Low	Moderate
	A5.127	SS.SCS.ICS.SLan	IGS.Lcon	Dense *Lanice conchilega* and other polychaetes in tide-swept infralittoral sand	Intermediate	High	Low	Moderate
	A5.223	SS.SSa.SSaVS.NintGam	IGS.NeoGam	*Neomysis integer* and *Gammarus* spp. in low salinity infralittoral mobile sand	Tolerant	NR	Not sensitive	High
	A5.231	SS.SSa.IFiSa.IMoSa	IGS.NcirBat	*Nephtys cirrosa* and *Bathyporeia* spp. in infralittoral sand	Low	Very high	Very Low	Moderate
	A5.272	SS.SSa.OSa.OfusAfil	CMS.AfilEcor	*Amphiura filiformis* and *Echinocardium cordatum* in circalittoral clean or slightly muddy sand	Intermediate	High	Low	Moderate

Habitat group	EUNIS code	2004 code	1997 code	Habitat name	Intolerance	Recovery	Sensitivity	Confidence
Slow growing epifauna								
	A4.1311	CR.HCR.XFa.ByErSp.Eun	MCR.ErSEun	Erect sponges, *%Eunicella verrucosa%* and *%Pentapora fascialis%* on slightly tide-swept moderately exposed circalittoral rock.	High	Very low	Very High	Moderate
	A4.2145	CR.MCR.EcCr.FaAlCr.Pom	MCR.FaAlC	Faunal and algal crusts, *Echinus esculentus*, sparse *Alcyonium digitatum* and grazing-tolerant fauna on moderately exposed circalittoral rock	High	Very high	Low	Low
Chalk reefs								
	A1.215	LR.MLR.BF.Rho	MLR.Rho	*Rhodothamniella floridula* on sand-scoured lower eulittoral rock	Intermediate	High	Low	Moderate
	A3.2113	IR.MIR.KR.Ldig.Pid	MIR.Ldig.Pid	*Laminaria digitata* and piddocks on sublittoral fringe soft rock	Intermediate	High	Low	Low
	A4.231	CR.MCR.SfR.Pid	MCR.Pid	Piddocks with a sparse associated fauna in upward-facing circalittoral very soft chalk or clay	Intermediate	Very high	Low	Low
	A4.232	CR.MCR.SfR.Pol	MCR.Pol	*Polydora* sp. tubes on upward-facing circalittoral soft rock	Intermediate	High	Low	Moderate
Vertical surfaces								
	A4.71	CR.FCR.Cv	CR.Cv	Caves and overhangs (deep)	Intermediate	Low	High	High
	A3.2112	IR.MIR.KR.Ldig.Bo	MLR.Fser.Fser.Bo	Underboulder communities	High	High	Moderate	Moderate
	A1.44	LR.FLR.CvOv	LR.Ov	Overhangs and caves	High	High	Moderate	High
	A3.715	IR.FIR.SG.CrSp	EIR.SCAn	Sponge crusts and anemones on wave-surged vertical infralittoral rock	High	High	Moderate	High

Appendix 2

List of parameters used in existing approaches to sensitivity assessment

Parameters			Reference
Physical	**Chemical**	**Biological**	
Mean maximum size of rhodolith Sediment structure			Bordehore et al. 2003
Area covered by maerl thalli Number of live maerl thalli Rate of sediment erosion			Hall-Spencer and Moore 2000b
% Maerl cover Mean dry weight of sediment			Hauton et al. 2003
1D (rugosity), 2D (area) and 3D (volume) heterogeneity of maerl			Kamenos et al. 2003
Species Range and distribution Species Substratum preferences		Adult/colony size range Age of sexual maturity Body flexibility Dispersal potential Environmental position Fecundity Feeding type Food Fragility Generation time Growth form/rate Larval settlement period Longevity/Life span Motility Reproduction (type/frequency/season) Sociability Typical abundance	Tyler-Walters et al. 2009
Trampling intensity from shore access to fishing grounds (duration/weight)			Cunningham et al. 1984, Tyler-Walters and Arnold 2008

Parameters			Reference
Physical	**Chemical**	**Biological**	
Tube growth			Vorborg 2000
Load bearing capacity			
		Population Density	Cranfield et al. 1999
Sediment texture		Population Density	Dolmer et al. 2001
Dissolved oxygen		% Mortality	Lenihan and Peterson 1998
Reef Height			
Salinity			
Water temperature			
Sediment structure		Density of post settlement juveniles	Piersma et al. 2001
		Population Density	
Species Range and distribution		Adult/colony size range	Tyler-Walters et al. 2009
Species Substratum preferences		Age of sexual maturity	
		Body flexibility	
		Dispersal potential	
		Environmental position	
		Fecundity	
		Feeding type	
		Food	
		Fragility	
		Generation time	
		Growth form/rate	
		Larval settlement period	
		Longevity/Life span	
		Motility	
		Reproduction (type/frequency/season)	
		Sociability	
		Typical abundance	
Gear type		Abundance	Kaiser et al. 2006
Habitat type (sand, muddy sand, mud, gravel, biogenic)		Biomass	
		Functional Group	
		Total number of species	
Depth of trawl		Population density	Bergman and Hup 1992

Parameters			Reference
Physical	**Chemical**	**Biological**	
Fishing effort		% Difference in density	Bergman and Van Santbrink 2000a,b
Depth of water within disturbed pits		Abundance	Dernie et al. 2003
Habitat recovery rate (mm day -1)		Number of species	
Sediment composition (organic content, % silt and clay content, water content)		Recovery rate (individuals/day)	
Sediment infilling rate			
Diet analysis		Abundance	Engel and Kvitek 1998
Fishing effort		Population density	
Sediment analysis		Species composition	
Gear type		Abundance	Ferns et al. 2000
		Bird activity (footprints per area)	
		Number of species	
		Population density	Kaiser et al. 1999
Gear type		Abundance	Kaiser et al. 2006
Habitat type (sand, muddy sand, mud, gravel, biogenic)		Functional Group	
		Total number of species	
Sediment organic concentration (carbon/nitrogen)		Abundance	Kenchington et al. 2001
		Biomass	
		Diversity	
		Species richness	
Grain-size analysis		Abundance	Pranovi et al. 2000
		Biomass	
		Mean Density of taxa	
Depth		Biomass	Queirós et al. 2006
Grain-size analysis		Production	
Trawling effort			
Burrowing rate		Size class	Robinson and Richardson 1998
Frequency of shell disturbance marks			
Abundance of organic matter in sediment			Schwinghamer et al. 1998
Depth of furrows			
Grain-size analysis			

Parameters			Reference
Physical	**Chemical**	**Biological**	
Species Range and distribution		Adult/colony size range	Tyler-Walters et al. 2009
Species Substratum preferences		Age of sexual maturity	
		Body flexibility	
		Dispersal potential	
		Environmental position	
		Fecundity	
		Feeding type	
		Food	
		Fragility	
		Generation time	
		Growth form/rate	
		Larval settlement period	
		Longevity/Life span	
		Motility	
		Reproduction (type/frequency/season)	
		Sociability	
		Typical abundance	
Fishing effort		Abundance	Veale et al. 2000
		Biomass	
		Diversity	
		Production	
		Species richness	
Depth in core		Abundance	Watling et al. 2001
Porosity		Biomass	
Sediment grain surface area			
Sediment organic composition (chlorophyll a, phaeopigment, carbon, nitrogen)			
Fishing effort		% Difference in density	Bergman and van Santbrink 2000a,b
Trawl door tracks		Biomass	Ball et al., 2000
		Diversity	
		Number of Individuals	
		Species richness	
Density and size of mounds		Abundance	Currie and Parry 1996

Parameters			Reference
Physical	**Chemical**	**Biological**	
Depth distribution of infauna Physical changes to seafloor % Total area dredged		Number of species	
Depth of water within disturbed pits Habitat recovery rate (mm day -1) Sediment composition (organic content, % silt and clay content, water content) Sediment infilling rate		Abundance Number of species Recovery rate (individuals/day)	Dernie et al. 2003
Gear type		Abundance Bird activity (footprints per area) Number of species	Ferns et al. 2000
Gear type Habitat type (sand, muddy sand, mud, gravel, biogenic)		Abundance Functional Group Total number of species	Kaiser et al. 2006
Niche Breadth		Abundance Diversity Population density	McConnaughey et al. 2000
Trawling disturbance		Biomass Body size distribution Production	Jennings et al. 2001
Gear type		Damage rate of cockles	Leitao and Gaspar 2007
Depth Grain-size analysis Trawling effort		Biomass Production	Queirós et al. 2006
Depth Fishing effort Sediment composition		Abundance Diversity Species richness	Schratzberger and Jennings 2002
Sediment composition (chlorophyll a)		Abundance Diversity Species richness	Sparks-McConkey and Wayling 2001

Parameters			Reference
Physical	**Chemical**	**Biological**	
Fishing effort		Abundance	Thrush et al.1998
Sediment composition (chlorophyll a)		Body size distribution	
		Diversity	
		Population density	
		Species richness	
Seabed topography		Abundance	Tuck et al. 1998
Sediment composition (organic carbon)		Biomass	
		Diversity	
		Number of species	
Species Range and distribution		Adult/colony size range	Tyler-Walters et al. 2009
Species Substratum preferences		Age of sexual maturity	
		Body flexibility	
		Dispersal potential	
		Environmental position	
		Fecundity	
		Feeding type	
		Food	
		Fragility	
		Generation time	
		Growth form/rate	
		Larval settlement period	
		Longevity/Life span	
		Motility	
		Reproduction (type/frequency/season)	
		Sociability	
		Typical abundance	
Fishing effort		Abundance	Bradshaw et al. 2000
		Diversity	
		Mean damage grade	
		Mortality (no killed/population)	
Depth		Abundance	Collie et al. 1997, 2000
Disturbance level		Biomass	
		Body size distribution	
		Diversity	

Parameters			Reference
Physical	**Chemical**	**Biological**	
		Number of species	
Factor intensity		Age of sexual maturity	McMath et al. 2000
		Growth form/rate	
		Recruitment	
Fishing effort		Population density	Craeymeersch et al. 2000
Fishing effort		Abundance by size class	Daan and Gislason 2005
		Average mortality	
		Maximum body length (Lmax)	
Fishing effort		Abundance of functional groups	de Juan et al. 2009
		Average size	
		Feeding type	
		Life form	
		Motility	
Depth of water within disturbed pits		Abundance	Dernie et al. 2003
Habitat recovery rate (mm day -1)		Number of species	
Sediment composition: (Organic content, % silt and clay, water)		Recovery rate (individuals day -1)	
Sediment infilling rate			
Grain-size analysis	Redox profile	Abundance	EMU 1992
		Diversity	
		Number of species	
Bottom stress	Dissolved Inorganic Nitrogen	Average trophic level	Fulton et al. 2005
Canyon coverage	Rate of nitrification and denitrification	Biomass	
Catch per unit effort (cpue)		Detrital dominance	
Depth		Diversity	
Erosion rate		Ecotrophic efficiency	
Light		Maximum body length (Lmax)	
No. Charismatic animals caught		Net Primary Production (NPP)	
Porosity		Potential biological removals (PBR)	
Salinity		Proportion of stock that are juveniles	

Parameters			Reference
Physical	**Chemical**	**Biological**	
Seabed type		Reproductive success (charismatic groups)	
Sediment composition (chlorophyll a)		Size at maturity	
System omnivory index (SOI)		Total mortality	
Temperature		Total system throughput	
		Trophic efficiency	
		Waste production	
		Abundance	Greenstreet and Rogers 2000
		Age at maturity	
		Growth form/rate	
		Length at maturity	
		Maximum body length (Lmax)	
Depth		Biomass	Heip et al. 1992
Latitude		Diversity	
Sediment composition (chlorophyll a)		Total biomass	
		Total density	
Fishing effort		Biomass	Hiddink et al. 2006
		Production	
Bed shear stress		Biomass	Hiddink et al. 2007
Erosion rate		Production	
Sediment composition (chlorophyll a)		% Mortality	
		Species richness	
		Typical abundance	
Catch rate		Density of taxa	Hoffman and Dolmer 2000
		Number of species	
Efficiency of capture (%)		Mean Damage Index (MDI)	Jenkins et al. 2001
		Species density	
		Abundance	Jennings et al. 1998
		Age at maturity	
		Length at maturity (Lmax)	
		Fecundity	
		Generation time	
		Mortality	
Season		Abundance	Kaiser et al. 1998
		Diversity	

Parameters			Reference
Physical	**Chemical**	**Biological**	
		Total individuals	
		Total species	
Number and volume of tube heads per sample		Individuals per tube head	Kaiser et al. 1999
		Species density	
Fishing effort		Abundance	Kaiser et al. 2000
		Biomass	
Gear type		Abundance	Kaiser et al. 2006
Habitat type (sand, muddy sand, mud, gravel, biogenic)		Functional Group	
		Total number of species	
Fishing effort		Body flexibility	MacDonald et al. 1996
Gear type		Body Strength	
		Larval supply	
		Larval settlement and recruitment success	
Catch number		Body size (thorax length)	Ramsay et al. 2000
Fishing effort		Diet composition (stomach contents)	
Catch rate		Abundance	Rogers and Ellis 2000
Mesh size		Maximum body length (Lmax)	
Fishing effort		Abundance	Rogers et al. 1999
Total catch number (%)		Biomass	
Total catch weight (%)		Diversity	
Bottom shear stress		Age at maturity	Tillin et al. 2006
Depth		Dissemination	
Fishing effort		Food	
Sediment composition: (Organic content, % silt and clay, water)		Feeding type	
		Habitat	
		Longevity	
		Mobility	
		Reproduction (frequency/technique)	
		Size (maximum wet weight)	
Grain-size analysis		Diversity	Thrush et al. 2001
Immobile biological features		Species richness	

Parameters			Reference
Physical	**Chemical**	**Biological**	
Mobile biological features			
Species Range and distribution		Adult/colony size range	Tyler-Walters et al. 2009
Species Substratum preferences		Age of sexual maturity	
		Body flexibility	
		Dispersal potential	
		Environmental position	
		Fecundity	
		Feeding type	
		Food	
		Fragility	
		Generation time	
		Growth form/rate	
		Larval settlement period	
		Longevity/Life span	
		Motility	
		Reproduction (type/frequency/season)	
		Sociability	
Geomorphological type of coast		Amphipod density	Weslawski et al. 1997
Ice cover type and duration		Bird moulting area in the intertidal zone	
Sediment flux		Export to sublittoral	
Substratum type		Haul out ground	
Stranded kelp on shore		Littoral supply from sublittoral	
Water transport/currents		Macrophyte cover	
Wave exposure		Recovery potential of intertidal	
Weathering potential		Resettlement potential	
		Seabird feeding ground	
		Species-specific vulnerability	
% Seagrass covering the seabed			Ardizzone et al. 2000
% Covering of dead matte			
Grain-size distribution			
Leaf area index (LAI)			

Parameters			Reference
Physical	**Chemical**	**Biological**	
Shoot density (n/m^2)			
Patch size		Abundance and density of associated fauna	Bell et al. 2001
Fishing effort		Biomass	Fonseca et al. 1984
Shoot number			
Patch size			Guillén et al. 1994
Trampling intensity from shore access to fishing grounds (duration/weight)		Biomass	Tyler-Walters and Arnold 2008
Patch size			Jackson et al. 2001
Area of impact		New patch recruitment (simulated recovery)	Neckles et al. 2005
Depth		Total Biomass	
% Canopy cover			
Distribution of patches			
Leaf width			
No. Leaves			
Shoot density (n/m^2)			
Shoot height (cm)			
Mean catch rate		Density of taxa	Freese et al. 1999
% Substrate disturbed		% taxa damaged	
Trampling intensity from shore access to fishing grounds (duration/weight)			Tyler-Walters and Arnold 2008
Species Range and distribution		Adult/colony size range	Tyler-Walters et al. 2009
Species Substratum preferences		Age of sexual maturity	
		Body flexibility	
		Dispersal potential	
		Environmental position	
		Fecundity	
		Feeding type	
		Food	
		Fragility	
		Generation time	
		Growth form/rate	
		Larval settlement period	

Parameters			Reference
Physical	**Chemical**	**Biological**	
		Longevity/Life span Motility Reproduction (type/frequency/season) Sociability Typical abundance	
Trampling intensity from shore access to fishing grounds (duration/weight)		Life form	Liddle 1991; Tyler-Walters and Arnold 2008
Aspect Latitude Longitude Soil Slope Trampling intensity from shore access to fishing grounds (duration/weight)		Life form Longevity Phenology Photosynthetic pathway Reproductive mode Root form	Yorks et al. 1997
Gear type		Abundance	Eno et al. 2001
Gear type		Abundance Assemblage biodiversity (specific phyla) Assemblage structure (specific phyla) Diversity	Kefalas et al. 2003
Habitat structure		Diversity	Thrush et al. 1998
Species Range and distribution Species Substratum preferences		Adult/colony size range Age of sexual maturity Body flexibility Dispersal potential Environmental position Fecundity Feeding type Food Fragility Generation time Growth form/rate Larval settlement period Longevity/Life span	Tyler-Walters et al. 2009

Parameters			Reference
Physical	**Chemical**	**Biological**	
		Motility Reproduction (type/frequency/season) Sociability Typical abundance	

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