The Marine Life Information Network® for Britain and Ireland (*MarLIN*)

Assessment of the Potential Impacts of Coasteering on Rocky Intertidal Habitats in Wales

Report to Cyngor Cefn Gwlad Cymru / Countryside Council for Wales

Contract no. NWR012

Dr Harvey Tyler-Walters

FINAL REPORT

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Contract specification

The relatively new recreational pursuit of Coasteering, which has developed in the St David's area of Pembrokeshire, appears to be expanding rapidly. The majority of local commercial recreation providers (outdoor pursuit centers etc) now appear to offer this pursuit. Coasteering has expanded out of St David's to other suitable cliff coastlines round Pembrokeshire. The majority of the rocky coastlines where it takes place lie within Pembrokeshire Marine Species Area of Conservation (SAC), and are also Sites of Special Scientific Interest (SSSI). No assessment has yet been undertaken of coasteering's potential impact on the intertidal habitats.

The contract specified a study of previous research/work into the impacts of physical disturbance through trampling / coasteering on rocky intertidal habitats, in particular work that may enable the following queries to be answered.

- What are the high risk communities?
- What are the vulnerable species? Are any of these important to the SAC / SSSI?
- Where are the high risk areas of coast? Do theses coincide with likely trampling / coasteering activities – i.e. where are the management priorities?
- Is there any similar research done related to climbing activities?
- Is there any difference in impacts on different shore types, both in terms of exposure and rock types?

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Assessment of the Potential Impacts of Coasteering on Rocky Intertidal Habitats in Wales

Executive summary

The relatively new recreational pursuit of Coasteering, which has developed in the St David's area of Pembrokeshire, appears to be expanding rapidly. The majority of local commercial recreation providers (outdoor pursuit centers etc.) now appear to offer this pursuit. The majority of the rocky coastlines where it takes place lie within Pembrokeshire Marine Special Area of Conservation (SAC), and are also Sites of Special Scientific Interest (SSSI). No assessment has yet been undertaken of coasteering's potential impact on the intertidal habitats. Therefore the Countryside Council for Wales (CCW) commissioned the Marine Life Information Network (*MarLIN*) to undertake a desk study of the likely environmental effects of coasteering on rocky intertidal habitats within the Pembrokeshire marine SAC.

The desk study was based on a review of the available literature, and in particular the effects of trampling on rocky intertidal communities. Communities (as biotopes) within the Pembrokeshire marine SAC likely to be exposed to coasteering activities were identified from Phase I biotope data for the area, provided by CCW. Where possible, existing research by *MarLIN* into the intolerance, recoverability and sensitivity of the biotopes identified, was used to identify their potential vulnerability to trampling.

The literature review revealed that:

- foliose canopy forming algae (e.g. fucoids) were particularly intolerant and sensitive to trampling impacts;
- trampling damaged erect coralline turfs, barnacles, and resulted in an increase in bare space; in some cases paths across the shore were visible;
- on brown algae dominated shores, understorey algae could suffer due to increased desiccation but algal turf species, opportunists and gastropod grazers (e.g. limpets) could increase in abundance as an indirect effect of trampling, and that
- trampling impacts resulted from physical contact and wear and were dependant on the intensity, duration, and frequency of trampling, and even the type of footwear used.

A total of 19 intolerant rocky intertidal biotopes were identified as potentially vulnerable to trampling and hence coasteering within the Pembrokeshire marine SAC, of which six are of Welsh importance and eight are nationally rare or scarce.

Trampling is a highly localized impact and it was not possible to identify biotopes, and hence communities, actually impacted by coasteering activities in the Pembrokeshire marine SAC. In addition, the majority of the literature addresses the impacts of trampling on wave sheltered or moderately exposed brown algal dominated shores, while coasteering occurs on more wave exposed, steeply inclined shores.

Therefore, direct survey of the routes used by coasteering groups within the Pembrokeshire marine SAC is required to identify the intensity, duration and frequency of trampling impact, together with the communities impacted. Given the paucity of data concerning trampling effects in the rocky intertidal in the UK, a survey of the impacts of coasteering would provide an opportunity to examine the effects of trampling and visitor use in steep rocky, wave exposed shores.

The report recognizes the potential to engage coasteerers in contributing to the development of strategies for minimizing adverse impacts, recording impacts and collecting information of use in identifying climate change and the occurrence of non-native species.

Rhwydwaith Gwybodaeth Bywyd Morol® Prydain ac Iwerddon (*MarLIN*)

Asesu Effeithiau Posib Arfordira ar Gynefinoedd Rhynglanw Creigiog yng Nghymru

Crynodeb Gweithredol

Gweithgaredd hamdden cymharol newydd sy'n ehangu'n gyflym yw arfordira, a ddatblygodd yn ardal Tyddewi yn Sir Benfro. Mae'r mwyafrif o ddarparwyr hamdden masnachol lleol (canolfannau gweithgareddau awyr agored ac ati) yn cynnig y gweithgaredd hwn. Mae'r mwyafrif o arfordiroedd creigiog a ddefnyddir o fewn Ardal Cadwraeth Arbennig (ACA) Forol Sir Benfro, ac yn Safleoedd o Ddiddordeb Gwyddonol Arbennig (SoDdGA) hefyd. Nid oes asesiad wedi'i wneud o effaith bosib arfordira ar gynefinoedd rhynglanw. Felly, gofynnodd Cyngor Cefn Gwlad Cymru (CCGC) i *MarLIN* gynnal astudiaeth ddesg o effeithiau amgylcheddol tebygol arfordira ar gynefinoedd rhynglanw creigiog yn ACA forol Sir Benfro.

Roedd yr astudiaeth ddesg yn seiliedig ar adolygiad o'r llenyddiaeth sydd ar gael, yn enwedig effeithiau sathru ar gymunedau rhynglanw creigiog. Cafodd y cymunedau (fel biotopau) a oedd yn debyg o gael eu defnyddio ar gyfer gweithgareddau arfordira yn ACA forol Sir Benfro eu nodi o ddata biotop Cam 1 ar gyfer yr ardal, a ddarparwyd gan CCGC. Lle bo'n bosib, defnyddiwyd ymchwil *MarLIN* i anoddefiad, y gallu i adfer a sensitifrwydd y biotopau a nodwyd i nodi eu natur agored posib i effeithiau sathru.

Dangosodd yr adolygiad o'r llenyddiaeth bod:

- algae deiliog sy'n ffurfio canopi (e.e. gwymon) yn anoddefgar a sensitif iawn i effeithiau sathru;
- sathru yn difetha tyweirch cwrelaidd unionsyth a chregyn llong, ac yn arwain at gynnydd mewn tir llwm; mewn rhai achosion roedd hi'n bosib gweld llwybrau ar draws y lan;
- ar lannau lle mae yna lawer iawn o algae brown, gallai'r algae oddi tano ddioddef yn sgil dysychiant ond gallai rhywogaethau mat o dyweirch, planhigion ymledol a phorwyr gastropodaidd (e.e. llygad maharen) gynyddu'n sylweddol fel effaith anuniongyrchol i sathru, a bod
- effeithiau sathru yn digwydd yn sgil cysylltiad corfforol a thraul ac yn dibynnu ar ddwyster, hyd ac amledd y sathru, a hyd yn oed y math o esgid roedd pobl yn eu gwisgo.

Nodwyd 19 o fiotopau rhynglanw creigiog anoddefgar fel rhai a allai gael eu heffeithio gan sathru ac arfordira yn ACA forol Sir Benfro, gyda chwech ohonynt o bwysigrwydd Cymreig ac wyth yn rhai sy'n brin yn genedlaethol.

Mae sathru yn effeithio ar rai mannau penodol ac nid oedd hi'n bosib nodi biotopau, nac felly gymunedau a oedd yn cael eu heffeithio'n uniongyrchol gan weithgareddau arfordira yn ACA forol Sir Benfro. Yn ogystal, mae mwyafrif y llenyddiaeth sydd ar gael yn sôn am effeithiau sathru ar lannau sy'n cael eu cysgodi rhag tonnau neu'n dod i gysylltiad cymedrol ag algae brown, tra bod arfordira'n digwydd ar lannau serth mwy agored i donnau.

Felly, mae angen gwneud arolwg uniongyrchol o'r llwybrau mae grwpiau arfordira yn eu defnyddio yn ACA forol Sir Benfro i nodi dwyster, hyd ac amledd y sathru, ynghyd â'r cymunedau sy'n cael eu heffeithio. O ystyried y diffyg data sydd ar gael ar effeithiau sathru yng nghynefinoedd rhynglanw creigiog y DU byddai arolwg o effeithiau arfordira yn gyfle i archwilio effeithiau sathru a defnydd ymwelwyr mewn glannau creigiog serth sy'n agored i donnau.

Mae'r adroddiad yn cydnabod y potensial i gynnwys arfordirwyr yn y gwaith o ddatblygu strategaethau i leihau effeithiau andwyol, i gofnodi effeithiau a chasglu gwybodaeth am ddefnydd o’r creigiau ar gyfer nodi newid yn yr hinsawdd a phresenoldeb rhywogaethau nad ydynt yn frodorol.

The Marine Life Information Network® for Britain and Ireland (*MarLIN*)

Assessment of the Potential Impacts of Coasteering on Rocky Intertidal Habitats in Wales

1. Aims and timetable

The relatively new recreational pursuit of Coasteering, which has developed in the St David's area of Pembrokeshire, appears to be expanding rapidly. The majority of local commercial recreation providers (outdoor pursuit centers etc.) now appear to offer this pursuit. Coasteering has expanded out of St David's to other suitable cliff coastlines around Pembrokeshire. The majority of the rocky coastlines where it takes place lie within the Pembrokeshire Marine Special Area of Conservation (SAC), and are also Sites of Special Scientific Interest (SSSI). No assessment has yet been undertaken of coasteering's potential impact on the intertidal habitats.

The contract aimed to study previous research / work into the impacts of physical disturbance through trampling / coasteering on rocky intertidal habitats, and in particular work that may enable the following queries to be answered.

- What are the high risk communities?
- What are the vulnerable species? Are any of these important to the SAC / SSSI?
- Where are the high risk areas of coast? Do these coincide with likely trampling / coasteering activities – i.e. where are the management priorities?
- Is there any similar research done related to climbing activities?
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The following review was carried out over a five day period between 7th and 25th of February 2005.

2. Methodology

The Biology and Sensitivity Key Information Sub-programme of the Marine Life Information Network (*MarLIN*) has reviewed the likely effects of a variety of marine activities on marine species and habitats in Britain and Ireland. As a result, the biology and sensitivity key information research represents a major review of relevant literature on the ecology of marine species and biotopes and the likely effects of human activities in the marine environment.

The *MarLIN* database and Web site presently holds information on the sensitivity (intolerance and recoverability) of over 152 marine species and 127 marine biotopes to changes in 24 separate environmental factors, including physical disturbance and abrasion. In particular, English Nature and Scottish Natural Heritage commissioned the biotope research and targeted priority biotopes identified within the interest features of UK marine SACs. In addition, the biotopes researched are used to represent another 157 biotopes within the 1997 biotope classification (version 97.06; Connor *et al.,* 1997a, b). Species and biotope research is detailed by Tyler-Walters *et al.* (2001) and Tyler-Walters & Hiscock (2003). *MarLIN*'s coverage of species and biotopes in Wales is detailed by Tyler-Walters *et al.* (2002; 2005).

2.1. Literature review

The *MarLIN* biology and sensitivity key information is based on the best available scientific literature. The following report was based on information previously researched by *MarLIN* together with an additional literature review of more obscure or recent material. The literature review was conducted using the resources of the National Marine Biological

Library, Plymouth and the University of Plymouth, together with relevant abstracting services such as the Aquatic and Fisheries Sciences Abstracts (ASFA), the ISI Web of Knowledge Service for UK Education, the National Information Services Corporation (NISC) Biblioline, and the British Library. However, only abstracts were available for some of the more obscure and low circulation reports. All references consulted are listed.

2.2. Identification of potentially vulnerable biotopes

The contract specified rocky intertidal habitats. Therefore, no attempt has been made to address any potential impacts on sedimentary, maritime cliff or coastal habitats (e.g. coastal grassland, salt marsh or sand dunes). Similarly, the potential disturbance of sea birds or sea mammals (e.g. seals and dolphins) has not been assessed.

The recent (2004) recreational audit of west Wales (Green *et al*., in prep.) identified 13 sites at which coasteering occurred (Figure 1). A list of biotopes and hence species, likely to be ‘exposed’ to coasteering was generated using Phase I biotope survey data for the Pembrokeshire marine SAC previously provided by the Countryside Council for Wales (Contract no FC 73-02-282; Tyler-Walters & Lear, 2004). Phase I biotopes that occurred within a 0.5 km radius of each site were identified using our in-house Geographical Information System (GIS). A radius of 0.5 km was chosen to ensure that all the biotopes that could be exposed to coasteering were chosen. Although Green *et al.* (in prep) specified sites at which coasteering occurred, no information on the exact parts of the site or paths used were available.

Additional biotopes were added from the SSSI citations for the relevant SSSIs, where available, and where GIS did not already identify them. The final list of potentially ‘exposed’ biotopes at each site is shown in Appendix 1. Please note that Phase I biotope data for Ceibwr Bay, Strumble Head, Aber-Eiddi and Lydstep Headland were not available.

Sedimentary biotopes were excluded from the list of potentially ‘exposed’ biotopes since they were not within the contract remit. However, likely sublittoral fringe biotopes were retained (see Section 3.4).

2.3. Sensitivity assessment

The sensitivity assessment rationale was developed by the *MarLIN* team in consultation with the Biology & Sensitivity Key Information Sub-programme Technical Management Group and ratified by the *MarLIN* programme Steering Group, both of which include representatives of the major users of marine information, statutory agencies, regulators, and marine research institutes. The *MarLIN* sensitivity assessment rationale, definitions of terms and scales used prior to March 2003 are given by Tyler-Walters *et al.* (2001) and their development in Tyler-Walters & Jackson (1999) and Hiscock *et al.* (1999). The definitions of sensitivity used after March 2003 are based on definitions suggested by the Review of Marine Nature Conservation (RMNC) (Laffoley *et al.*, 2000) and developed by *MarLIN* in consultation with our Biology & Sensitivity Key Information Sub-programme Technical Management Group and Sensitivity Mapping Advisory Group. No attempt has been made to reproduce their information here, except by way of explanation. The reader should refer to the above reports or the *MarLIN* Web site for detailed information.

Terms such as ‘intolerance’, ‘recoverability’ ‘sensitivity’ and ‘vulnerability’ are often used to express a variety of meanings. Therefore, the standard definitions and terms used in the following report are shown in Box 1. The sensitivity assessment rationale for species and biotopes is summarized in Appendix 2 and notes on the interpretation of *MarLIN* sensitivity assessments in Appendix 3.

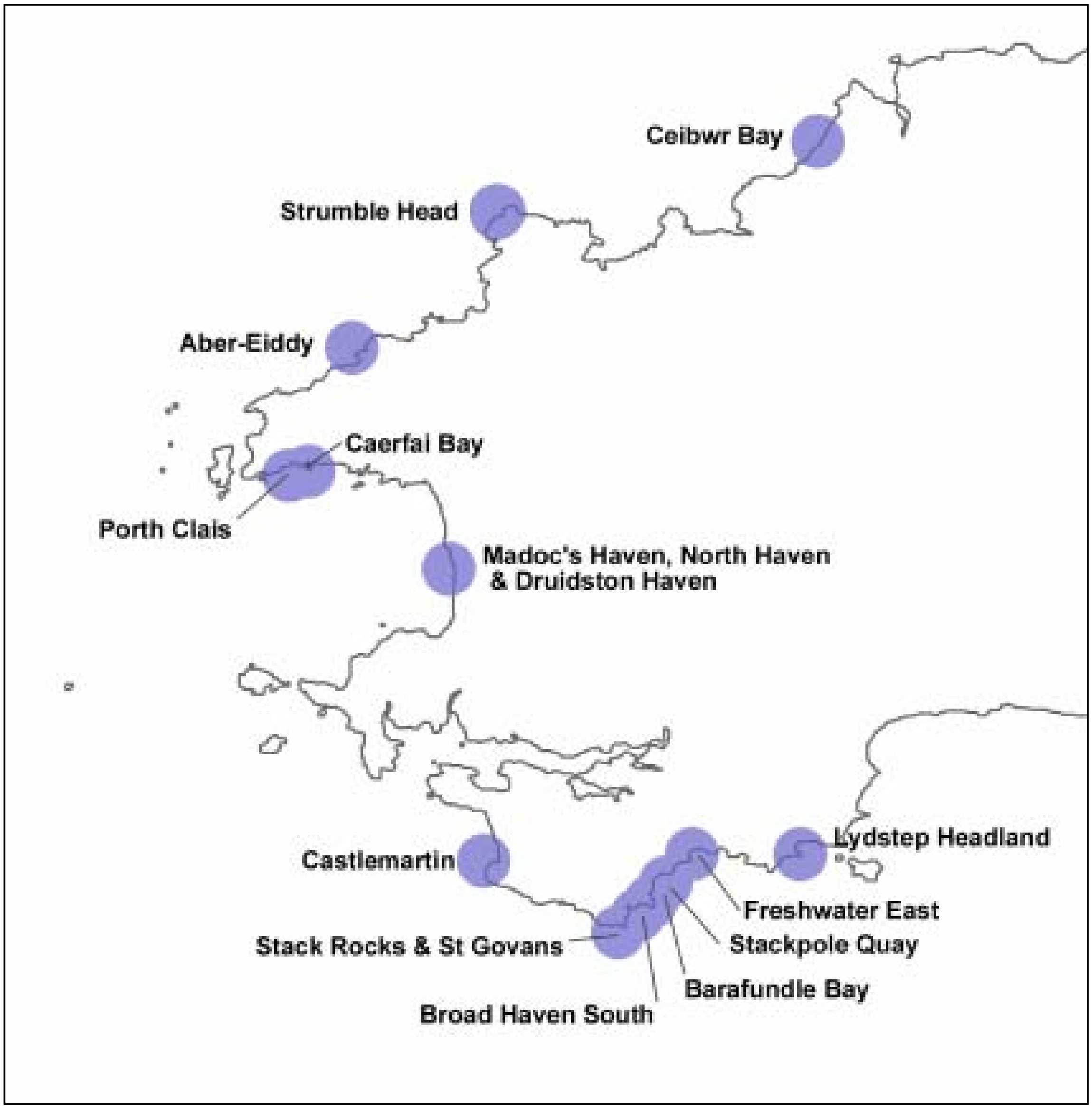

Figure 1. Sites within west Wales where coasteering is known to occur (Green *et al.,* in prep).

3. Results of the literature review

The ecological impacts of outdoor recreation have been studied in detail in terrestrial and aquatic environments under the theme of 'recreational ecology' (Liddle, 1997). The mechanical forces and wear exerted by trampling by humans and animals, different modes of transport (e.g. trail-bikes, cars, and four-wheel-drive vehicles), camping and boating are reviewed by Liddle (1997). The effects of trampling on terrestrial plant communities, e.g. in salt marsh and sand dunes communities are well studied. Trampling has been shown to cause the decline in the height, cover and biomass of plants with an increasing trampling intensity. Some species are more resistant or tolerant than others, and the disturbance may cause an initial increase in the cover of some species (Liddle, 1991). However, intensive trampling eventually results in bare space or bare paths, and can cause cumulative erosion and soil compaction (Liddle, 1997).

In plants, small size, folded leaves, rosette habit, a growth form that protects the meristem from damage, and small cell size have been identified as resistant features (Liddle, 1991, 1997). Plants can also be grouped into susceptibility categories dependant on the likelihood of damage and their rates of recovery, in a similar manner to sensitivity.

Box 1. Standard definitions

'Biotope' refers to the combination of physical environment (habitat) and its distinctive assemblage of conspicuous species. For practical reasons of interpretation of terms used in directives, statutes and conventions, in some documents, 'biotope' is sometimes synonymized with 'habitat'.

'Habitat' the place in which a plant or animal lives. It is defined for the marine environment according to geographical location, physiographic features and the physical and chemical environment (including salinity, wave exposure, strength of tidal streams, geology, biological zone, substratum), 'features' (such as crevices, overhangs, or rockpools) and 'modifiers' (for example sand-scour, wave-surge, or substratum mobility).

'Community' refers to a group of organisms occurring in a particular environment, presumably interacting with each other and with the environment, and identifiable by means of ecological survey from other groups. The community is usually considered the biotic element of a biotope.

'Intolerance' is the susceptibility of a habitat, community, or species (i.e. the components of a biotope) to damage, or death, from an external factor. Intolerance must be assessed relative to specified change in a specific environmental factor.

'Recoverability' is the ability of a habitat, community, or species (i.e. the components of a biotope) to return to a state close to that which existed before the activity or event caused change.

'Sensitivity' is dependent on the intolerance of a species or habitat to damage from an external factor and the time taken for its subsequent recovery. For example, a "highly sensitive" species or habitat is one that is very adversely affected by an external factor arising from human activities or natural events (killed/destroyed, 'high' intolerance) and is expected to recover only over a very long period of time, (10 to 25 years: 'low' recoverability). Intolerance and hence sensitivity must be assessed relative to a specified change in a specific environmental factor.

'Vulnerability' expresses the likelihood that a habitat, community or species will be exposed to an external factor to which it is sensitive. Degree of 'vulnerability' therefore indicates the likely severity of damage should the factor occur at a defined intensity and/or frequency.

Similarly, the growth form of corals was found to influence the level of damage inflicted by visitors walking across coral reefs in the Great Barrier Reef. Digitate, wedge or blade like, encrusting and massive forms were tolerant of trampling, while plate, foliaceous and open arborescent forms were intolerant (Liddle, 1997). Again, the species could be categorized by their resistance to damage and ability to recover. For example, resilient forms were defined as species with a low resistance to damage but with high recovery rates (Liddle, 1997).

3.1. The potential effects of coasteering in the rocky shores

Coasteering is a recent development in recreation, which attracts a wide variety of people, from those wishing to explore otherwise inaccessible environments to people seeking the thrill of jumping into, and riding, waves and surf. Coasteering combines sea level traversing, scrambling, surf swimming and cliff jumping to produce a blend of climbing, scrambling and swimming to access caves, plunge pools and jets of raging white water (South West Watersports, 2004). South West Watersports (2004) state "it [coasteering] encapsulates a shoreline adventure where you can experience the forces of the sea at first hand and visit those remote places only frequented by the spirited and the brave".

In practice, coasteering involves walking to the top of rocks, steep rocky inclines, cliffs and gulleys, and then jumping into the waters below. Participants (henceforth 'coasteerers') may then swim along to other parts of the shore and/or then scramble over the rock surfaces at the bottom of the shore as they leave the water on route to another rock surface or gully. Therefore, impacts on the rocky intertidal habitats present stem from direct physical contact between people and the species present. The following types of physical contact and hence physical disturbance can be envisaged.

1. Physical contact and knocking of gully walls while jumping into the water, or by being pushed against the walls by wave action.
2. Brushing against and touching of epifaunal crusts and turfs on overhangs and vertical surfaces, while swimming between parts of the shore.
3. Collection of souvenir organisms.
4. Pulling on seaweeds, especially kelps in the lower shore and sublittoral fringe, as handholds while scrambling out of the water.
5. Trampling over rock surfaces, whilst walking between parts of shore, climbing inclines, or scrambling out of the water.
6. Trampling on rock surfaces while waiting in turn to jump, i.e. pivoting and waiting in one spot.

Physical contact with gully walls is likely to be similar to the impact caused by wave driven debris (e.g. logs), resulting in crushing and removal of epifauna. However, it is likely to be a minor consideration in coasteering since it is potentially detrimental or injurious to the coasteerers themselves. Similarly, brushing against epifaunal crusts and turfs is likely to be minor and probably less severe than results from wave driven debris. Collection of organisms as souvenirs of the experience is not the main focus of the event, and no evidence for this activity within coasteering was found.

Pulling on seaweeds in the lower littoral and sublittoral fringe is likely to result in removal of specimens, especially the smaller species, e.g. turf forming species. But coasteerers are likely to avoid species that come away in their hands. Kelps are more robust but pulling on stipes may result in breakage of the stipe or damage to the growing meristem, resulting in loss of the affected plants. Occasional loss of kelp plants due to grazing, wave action and age is part of the dynamic nature of kelp beds, although continuous damage to plants in one particular position on the shore is likely to result in loss of the kelps at that position (see Section 3.4).

Trampling has been shown to be an additional type of physical disturbance on rocky shore habitats, and the pre-adaptation of macroalgae and sessile organisms to wave action does not necessarily provide protection or tolerance of the effects of trampling. Brosnan & Cumrine (1994) noted that storms and wave driven logs resulted in localized and seasonal (winter) disturbances often resulting in patches of bare space. Trampling also resulted in bare space in some communities but was likely to be chronic in nature and more frequent in spring and summer (less so in winter). They noted that many species are adapted to take advantage of bare space left by winter storms, and peak recruitment for many species (e.g. algae and barnacles) occurs in spring and summer, which coincides with peak periods for visitation of shores, and hence trampling (Brosnan & Cumrine, 1994).

Therefore, the following report concentrates on trampling and other physical disturbance impacts. The report reviews current literature on the impacts of trampling on specific rocky shore species and habitats, and its relation to coasteering and then identifies species and habitats that may be vulnerable as a result.

3.2. Nature of the information

Although, the recreational ecology of terrestrial and some coastal habitats is well studied, there are relatively few studies of the effects of trampling on rocky shore communities. The type of study, location and habitat examined, and trampling intensities, are summarized in Table 1. The majority of studies were conducted overseas and the affected species do not occur in UK waters. Study techniques also varied, from comparative studies of sites with visitors to those without, to careful experimental studies with varying degrees of trampling intensity. In addition, the shores examined tended to be shores that were subject to or threatened by recreational use, and therefore tended to be shores that were easily accessible.

3.3. The effects of trampling on intertidal rocky shore communities

While the species examined in many of the studies reviewed may not occur in the UK, the communities examined have counterparts on UK shores. The effects of trampling on the following community types have been reported:

- brown algal mats or brown algal shrubs;
- algal turfs;
- coralline turfs (characterized by *Corallina* spp.);
- the barnacle zone;
- mussel beds;
- Sabellariid worms, and
- shallow infralittoral algal communities.

In addition, limpet and other gastropod grazers were examined.

3.3.1. Brown algal shrubs

The majority of the studies cited (see Table 1) examined the effects of trampling on brown algal shrubs or mats, i.e. mats of canopy forming, fleshy, brown algae.

In Australia, the articulated brown algae *Hormosira banksii* was reported to be severely affected by trampling (Povey & Keough, 1991; Keough & Quinn, 1998; Schiel & Taylor, 1999). Povey & Keough (1991) observed a 50% reduction in *H. banksii* cover within 12 days of high intensity trampling (25 passes/tramples per day), and paths became visible in the brown algal mats within four days of trampling. After ca 6 weeks (includes 12 days of trampling), transects were clear of *H. banksii.* Low intensity trampling (two passes/tramples per day) reduced *H. banksii* cover and paths were visible after ca 6 weeks trampling, although considerable cover of *H. banksii* remained. After 270 days, the low intensity treatments recovered by growth from existing holdfasts, while *H. banksii* cover was still <50% of controls in high intensity treatments. After a further 150 days, the high intensity treatments reached 50% cover, which was markedly less than controls (Povey & Keough, 1991). The fronds of *H. banksii* are composed of rows of articulated vesicles, which may make it particularly susceptible to trampling damage. Povey & Keough (1991) noted that a single step could remove up to 34% of the frond, as pieces are easily broken off. Fletcher & Frid (1996a) noted that the low trampling intensity used above is equivalent to as few as two visitors per day walking across the transect.

Table 1. Summary of characteristics of studies cited. The type of habitat and degree of wave exposure are expressed as described in the papers cited. (?) = wave exposure was not indicated and has been inferred from the communities present. ? = characteristic unknown.

Study	Location	Habitat type	Shore type (wave exposure)	Community type examined	Type of study	Trampling intensity (period/duration)
Bally & Griffiths (1989)	Dalebrook, Cape Town, South Africa	Gently sloping sandstone	Moderately exposed to exposed (?)	Littorinid zone, Barnacles (balanoid zone)	Experimental: 4x 31 m transects plus one single trampling point experiment.	0,10,100, & 500 times /month (3 months)
Beauchamp Gowing (1982)	Santa Cruz, California	Rock platforms (?)	Moderately exposed to exposed (?)	Mussels beds Barnacles, Brown algal mats	Comparative: 20x 0.1 m^2 quadrats at 3 sites of low, intermediate and high visitor use.	0, 1 and 7 people/day depending on site (autumn and spring).
Boalch *et al.* (1974); Boalch & Jephson (1981)	Wembury, Devon, UK	'Slatey' undulating rocky shore	Moderately exposed	Brown algal mats	Resurvey: Resurvey of Colman's 1931 transects.	None
Brosnan (1993); Brosnan & Crumrine (1994)	Newport, Oregon, USA	Flat basaltic benches	Moderately exposed (?)	Brown algal mats Algal turf Barnacles Mussel bed	Experimental: Trampling – 0.2x0.2 m (algae) or 0.2x0.3 m (mussels) blocks. Human exclusion.	Blocks trampled 250/month (12 months).
Brown & Taylor (1999)	Cape Rodney to Okakari Point Marine Reserve, New Zealand	Intertidal reef flat	Moderate to exposed (?)	Coralline algal turf	Experimental: 4 x 0.09 m^2 quadrats.	0, 2, 5, 30 footsteps/day (5 days).

Study	Location	Habitat type	Shore type (wave exposure)	Community type examined	Type of study	Trampling intensity (period/duration)
Denis & Murray (2001).	South California	?	Moderately exposed (?)	Brown algal mats.	Experimental: 15 0.5x0.7 m blocks.	0, 150, or 300 foot steps / month (16 months).
Erickson *et al.* (2004)	Olympic National Park, Washington	?	Moderately exposed (?)	Brown algal mats, Mussel beds, Barnacles.	Comparative: Sites accessible to visitors vs. inaccessible sites.	Not specified.
Fletcher & Frid (1996a)	Cullercoats Bay & St. Mary's Island, Newcastle upon Tyne, UK	Flat sandstone shore	Moderately exposed	Brown algal mats.	Experimental: 2 sites, 4x 1 m^2 blocks.	0, 20, 80, 160, footsteps/ m^2 per spring tide (9 months).
Fletcher & Frid (1996b)	Cullercoats Bay & St. Mary's Island, Newcastle upon Tyne, UK	Flat sandstone shore	Moderately exposed	Brown algal mats.	Experimental: 2 sites, 4x 1 m^2 blocks.	0, 20, 80, 160, footsteps /m^2 per spring tide (16 months).
Ghazanshahi *et al.* (1983)	Palos Verdes Peninsula, S California, USA	Gentle rocky slopes and low reef	Moderate to exposed (?)	Barnacles, Algal turf, Coralline algae, Sabellariid worms.	Comparative: Survey of 20 m transects at 13 sites of different visitor intensity.	High = >1.7 persons/10 m/day Low- <1.3 persons/100 m/day
Jenkins *et al.* (2002) (abstract only)	San Juan Country Park, Washington, USA	?	Sheltered to moderately exposed (?)	Brown algal mats.	Experimental: 6 3-5m transects.	250 steps/transect, 3 times /week for 6 weeks.
Keough & Quinn (1991)	Review article	N/A	N/A	N/A	Review: Discussed past and present work by authors and others.	See Povey & Keough (1991).

Study	Location	Habitat type	Shore type (wave exposure)	Community type examined	Type of study	Trampling intensity (period/duration)
Keough & Quinn (1998)	Mornington Peninsula National Park, SE Australia	Flat limestone platforms	Moderate exposed to sheltered	Brown algal mats, Coralline algal mats.	Experimental: 0.5x2m transects.	0, 2 & 25 passes /day, using rubber-soled shoes (6-8 days /summer for 6 years)
Milazzo *et al.* (2002)	Ustica Island, W Mediterranean, Italy	Horizontal basaltic platform	Sheltered	Shallow infralittoral algal turfs.	Experimental: 18 0.4x2m transects.+	0, 10,25,50,100, & 150 passes of transect, using gumboots.
Murray *et al.* (2001)	Orange & Los Angeles counties, California	?	?	Macroalgae.	Resurvey: Comparison of recent survey results to surveys in the 1950s, 60s, 70s, and 80s.	Not identified.
Povey & Keough (1991)	Mornington Peninsula National Park, SE Australia	Flat limestone platforms	Moderate exposed to sheltered	Brown algal mats, Coralline algal mats, Bare rock, Mussel beds.	Experimental: Single steps, gastropod dislodgement, kicking/stepping on limpets, and 0.5x2 m transects (every daytime low tide from July-October)	Transects: 0, 2 & 25 passes/day, using rubber-soled shoes Small scale effects: 1, 10 50 or 75 steps (single tide)
Schiel & Taylor (1999)	Wairepo flats, South Island, New Zealand	Gently sloping siltstone platforms	Sheltered to moderately exposed (?)	Brown algal mats.	Experimental: 7x 0.3mx2 m transects. Trampling initiated in spring and autumn.	0, 10, 25, 50, 100, 150 & 200 tramples

Schiel & Taylor (1999) noted that as few as 10 tramples reduced cover by of *H. banksii* by 25% in New Zealand, while >90% of cover was removed by 200 tramples. Most damage occurred within the first one to two months of trampling. Damage increased with increasing trampling intensity. At one of their sites the plants were easily dislodged because the substratum was composed of soft siltstone. Recovery was dependant on season. Treatments carried out in spring regained 50% cover within 5 months, and control levels within 21 months. However, recovery in autumn treatments was delayed until recruitment in the following summer (ca 11 months) and recovery was still proceeding after 16 months (Schiel & Taylor, 1999).

Keough & Quinn (1998) examined the effects of different trampling intensities on rocky shore communities over a six year period. The experiment involved 6-8 days trampling per transect at 0, 5, 10 or 25 passages per trampling, every summer for 6 years. The effects of trampling varied with site. At one site, trampling resulted in a reduction in cover, proportional to the trampling intensity. Recovery occurred by the following summer but an even greater decline was seen in the next summer, with little subsequent recovery and the intermediate treatments remained at 60-70% cover. High intensity trampling, however, caused a severe decline, with little recovery and after four years cover remained <10%. At another two sites, trampling resulted in an initial decline and recovery (within 8-9 months) and subsequent greater decline as above. But all plots recovered completely and no trampling effects were observed over the next 3 years. Keough & Quinn (1998) suggested that there was greater variation in trampling effects between sites than within treatments but did not determine the cause of the variation.

Murray *et al.* (2001) resurveyed southern California shores previously surveyed in the 1950s, 60s, 70s, and 80s. They reported a decrease in fleshy macrophyte cover and diversity, with increases in crustose and articulated (erect) coralline algae and small turf-forming algal species. They suggested that the rocky shore community changes were due to an increase in coastal development and the resident human population, although they did not distinguish between recreational use and pollution effects.

Beauchamp & Gowing (1982) compared rocky shore communities between sites that varied in visitor use on the California coast. They noted a general pattern of higher diversity and density of species at the less trampled sites. Most noticeable was the absence of the brown alga *Pelvetiopsis limitata* at the most trampled site. In a comparative survey of low and high use sites in southern California, Ghazanshahi *et al.* (1983) noted that the overall algal abundance ‘rank’ was lower where public use was higher. However, their abundance rank combined foliose and turf forming algal species.

On the coast of Oregon, Brosnan (1993) reported a significant reduction in brown foliose algae (the fucoids *Pelvetiopsis limitata* and *Fucus distichus*, and foliose red alga *Iridaea cornucopiae*) as a result of trampling (250 tramples per plot for one day per month for 12 months). Their abundance were reduced from 80% to 35% within a month of the start of trampling, and remained so for the rest of the experiment. In a visitor exclusion experiment, foliose algae increased from 62% to 94.5% cover in six months. When visitor access was returned foliose algae declined rapidly.

Brosnan & Crumrine (1994) noted that trampling significantly reduced algal cover within 1 month of trampling. Foliose algae were particularly affected and decreased in cover from 75% to 9.1% in trampled plots. *Mastocarpus papillat*us decreased in abundance from 9% to 1% in trampled plots but increased in control plots. *Fucus distichus* decreased in the summer months only to recover in winter but in trampled plots remained in low abundance (between 1 and 3% cover). Trampling resulted in a decrease in cover of *Pelvetiopsis limitata* from 16% to 1.5%. *Iridaea cornucopiae* decreased from 38 to 14% cover within a month and continued to decline to 4-8% cover. However, after trampling ceased, recovery of algal cover including *Iridaea cornucopiae* and *Mastocarpus papillat*us was rapid (ca 12 months) (Brosnan & Crumrine, 1994).

On the Washington coast, experimental trampling reduced the cover of *Fucus gardneri* by 30% within 6 weeks and remained low for a further 3 months (Jenkins *et al.,* 2002). Similarly, trampling resulted in decreased vegetative and reproductive biomass, and reduced size in the fucoid *Silvetia compressa* in southern California (Denis & Murray, 2001).

In the UK, Boalch *et al.* (1974) and Boalch & Jephson (1981) noted a reduction in the cover of fucoids at Wembury, south Devon, when compared to surveys conducted by Colman (1933). The size ranges of *Ascophyllum nodosum, Fucus vesiculosus* and *Fucus serratus* were skewed to smaller length, and the abundance of *A. nodosum* in particular was reduced (Boalch & Jephson, 1981). It was suggested that visitor pressure, especially after the construction of a car park, was responsible for the reduced cover of fucoids (Boalch *et al.*, 1974). They suggested that the raised edges of the slatey rock severed fronds when the rocks were walked over. However, no quantitative data was provided.

Fletcher & Frid (1996a, b) examined the effects of persistent trampling on two sites on the north east coast of England. The trampling treatments used were 0, 20, 80, and 160 steps per m^2 per spring tide for 8 months between March and November. Using multivariate analysis, they noted that changes in the community dominated by fucoids (*Fucus vesiculosus, F. spiralis* and *F. serratus*) could be detected within 1 to 4 months of trampling, depending on intensity. Intensive trampling (160 steps/m^2/spring tide) resulted in a decrease in species richness at one site. The area of bare substratum also increased within the first two months of trampling but declined afterwards, although bare space was consistently most abundant in plots subject to the greatest trampling (Fletcher & Frid (1996a, b). The abundance of fucoids was consistently lower in trampled plots than in untrampled plots. Fletcher & Frid (1996a) noted that the species composition of the algal community was changed by as little as 20 steps per m^2 per spring tide of continuous trampling since recolonization could not occur. A trampling intensity of 20 steps per m^2 per spring tide could be exceeded by only five visitors taking the same route out and back again across the rocky shore in each spring tide. Both of the sites studied receive hundreds of visitors per year and damage is generally visible as existing pathways, which are sustained by continuous use (Fletcher & Frid, 1996a, b). However, the impact was greatest at the site with the lower original abundance of fucoids.

In summary brown algal shrubs, characterized by fucoid algae (*Fucus* spp. in the UK), are particularly intolerant of trampling, depending on intensity. Fucoid algae demonstrate a rapid (days to months) detrimental response to the effects of trampling, depending on species, which has been attributed to either the breakage of their fronds across rock surfaces (Boalch *et al.*, 1974) or their possession of small discoid holdfasts that offer little resistance to repeated impacts (Brosnan & Crumrine, 1992; cited in Fletcher & Frid, 1996b). Brosnan (1993) suggested that the presence or absence of foliose algae (e.g. fucoids) could be used to indicate the level of trampling on the rocky shores of Oregon.

3.3.2. Algal turfs

Brosnan (1993) noted that algal turf species (*Endocladia muricata* and *Gelidium* spp.) increased by 38% in trampled plots as foliose algae declined, and algal turf dominated trampled areas. Exclusion of visitors, and hence reduced trampling, reduced relative algal turf abundance by 31%, while foliose algae increased in abundance. Brosnan & Crumrine (1994) noted that the algal turf forming species *Endocladia muricata* showed the least change in cover as a result of trampling, from 5% to between 3 and 5%. *Endocladia muricata* recovered quickly after trampling ceased and increased its cover to 5.6%, slightly higher than before trampling. Similarly, Jenkins *et al*. (2002) noted that *Endocladia muricata* did not decline significantly in response to trampling.

Fletcher & Frid (1996a, b) reported a decrease in the understorey algal community of encrusting coralline algae and red algae, which was probably an indirect effect due to increased desiccation after removal of the normally protective fucoid canopy (see Hawkins & Harkin, 1985) by trampling. They also noted that opportunistic algae (e.g. *Ulva* sp.) increased in abundance. Schiel & Taylor (1990) also observed a decrease in understorey algae (erect and encrusting corallines) after 25 or more tramples, probably due to an indirect effect of increased desiccation as above. However, Schiel & Taylor (1999) did not detect any variation in other algal species due to trampling effects. Similarly, Keough & Quinn (1998) did not detect any effect of trampling on algal turf species.

In summary algal turfs seem to be relatively tolerant of the direct effects of trampling (based on the available evidence) and some species may benefit from removal of canopy forming algae. Their tolerance may result from their growth form as has been shown for vascular plants and corals (Liddle, 1997). Brosnan (1993) suggested that algal turf dominated areas (on shores usually dominated by fucoids) were indicative of trampling on the rocky shores of Oregon However, tolerance is likely to vary with species and their growth form and little species specific data was found. Furthermore, algal turf may suffer negative indirect effects where they form an understorey below canopy forming species.

3.3.3. Coralline algal turfs

Erect coralline algae (e.g. *Corallina* spp.) can form extensive turfs in wave exposed conditions, or in shallow rocky pools, that harbour a diverse array of amphipods and meiofauna, and support a variety of red algae. The effect of trampling on erect coralline algal turf in New Zealand was studied by Brown & Taylor (1999). For example, moderate trampling (50 steps per 0.09 m^2) or more reduced turf height by up to 50%, and the weight of sand trapped within the turf to about one third of controls. This resulted in declines in the densities of the meiofaunal community of gastropods, ostracods, and polychaetes within two days of trampling. The community returned to normal levels (except polychaetes) within 3 months of trampling events (Brown & Taylor, 1999). However, their experiment only subjected the turf to five days of trampling.

Zedler (1976, 1978; cited in Gharanshahi *et al.,* 1983) reported a reduction in coralline algae abundance in areas of Cabrillo National Monument, San Diego, subject to heavy visitor use, and further noted that coralline algae decreased when visitor use increased. Povey & Keough (1991) noted that erect coralline turf was damaged by intensive trampling and was reduced in height by 50% compared to other treatments (low intensity and control). In addition, while the overall cover of coralline turf increased by 11% in other treatments, it only rose by 3% in transects trampled at high intensity but no significant effect on cover was seen at the end of the trampling experiment (Povey & Keough, 1991).

Fletcher & Frid (1996a, b) noted a decrease in the understorey algal community of encrusting coralline algae and red algae, which was probably an indirect effect due to increased desiccation after removal of the normally protective fucoid canopy (see Hawkins & Harkin, 1985) by trampling. Similarly, Schiel & Taylor (1999) noted that trampling had a direct detrimental effect on coralline turf species on the New Zealand rocky shore. At one site, coralline bases were seen to peel from the rocks (Schiel & Taylor, 1999), although this was probably due to increased desiccation caused by loss of the algal canopy. Keough & Quinn (1998) also noted a slight (8%) decrease in erect coralline turf cover in their most intensive trampling, at one site only. However, again this may have been due to increased desiccation.

In summary erect coralline turf is probably of intermediate intolerance of trampling, demonstrating a reduction in turf height and reduced cover in the highest trampling intensities studied. Brown & Taylor (1999) noted that a reduction in turf height was due to tissue loss. The resident meiofaunal community is intolerant but recovers quickly, and associated foliose red algae (e.g. *Mastocarpus papillat*us) are also probably intolerant.

3.3.4. Barnacle zone

Jenkins *et al.* (2002) did not observe any effects on barnacle cover as a result of trampling. Similarly, Beauchmap & Gowing (1982) did not observe any difference in barnacle density between sites with different levels of visitor use. However levels of visitor use (trampling intensity) were low in comparison with other studies (Table 1). Bally & Griffiths (1989) listed the removal of dead barnacles as one of the immediate effects of trampling but did not observe any longer term effects in any fauna, although their study is unique in this respect (see Section 3.4).

Ghazanshahi *et al.* (1983) reported that *Balanus glandula* exhibited reduced cover at all shore heights with increasing public use, and suggested that trampling rather than collecting was the likely cause. However, cover in this species varied between ca 0.1% and 1.5%. Keough & Quinn (1991) and Ghazanshahi *et al.* (1983) cited studies by Zedler (1978) that suggested that barnacles and polychaetes decreased in abundance with increased public use. Erickson *et al.* (2003) found that visitor accessible areas of Olympic National Park coast had a greater percentage cover of bare space in five of seven sites examined. They observed significantly greater numbers of *Balanus glandula* barnacle scars (remains of bases when a barnacle is removed or dies) in accessible areas, and noted that barnacles were consistently smaller in more accessible areas. However, they did not detect any significant differences between treatment and reference sites in their pilot study.

However, Brosnan & Crumrine (1994) reported that trampling significantly reduced barnacle cover at both of their study sites, falling from 66.6% to 7.2% in 4 months at one site and from 21.3 to 5.1% within 6 months at the other. Cover remained low until recruitment in the following spring. Similarly, barnacle cover as epibionts on mussels was reduced significantly in the first month following trampling. Overall, trampling crushed barnacles on rocky or mussel substrata. In single step experiments, *Chthamalus antennatus* were the most easily crushed species, and about 15% of individuals were crushed by a single step, while less than 5% of littorinids and mussels were crushed (Povey & Keough, 1991). Nevertheless, Brosnan & Crumrine (1994) noted that decreased algal cover due to trampling could increase bare space for settlement by barnacles.

In summary the effects of trampling on barnacles seems to be variable, with some studies not detecting significant differences between trampled areas and controls. However, in the case of Beauchamp & Gowing (1982) trampling intensity was low, while Ghazanshahi *et al.* (1983) examined low abundance populations. The worst case incidence was reported in the algal-barnacle assemblage studied by Brosnan & Crumrine (1994), which may be more representative of barnacle dominated shores. Overall, barnacles are probably relatively easily damaged and crushed by trampling, and are regularly heard to ‘crunch’ under foot while walking on the shore.

3.3.5. Mussel beds

Brosnan & Crumrine (1994) reported large declines of mussels (*Mytilus californianus*) from mussel beds due to trampling. On a single day 54% of mussels were lost from a single experimental plot. Mussels continued to be lost throughout the experimental period, forming empty patches larger than the experimental plots. The empty patches continued to expand after trampling had ceased, due to wave action. At another site, the mussel bed was composed of two layers, so that while mussels were lost, cover remained. Brosnan & Crumrine (1994) suggested that trampling destabilizes the mussel bed, making it more susceptible to wave action, especially in winter. Recruitment within the plots did not occur until after trampling had ceased, and no recovery had occurred within 2 years. Brosnan (1993) also reported a 40% loss of mussels from mussel beds after one month of trampling, and a 50% loss within a year.

Brosnan & Crumrine (1994) noted that mussels that occupied hard substrata but did not form beds were adversely affected. Although only at low abundance (2.5% cover), all mussels were removed by trampling within 4 months. Brosnan & Crumrine (1994) noted that in earlier experiments mussels were not common and confined to crevices in heavily trampled sites. Similarly, the mussel beds infauna (e.g. barnacles) was adversely affected, and were crushed or lost with the mussels to which they were attached. However, Beauchamp & Gowing (1982) did not observe any differences in mussel density between sites that differed in visitor use.

In summary trampling is likely to destabilize mussel beds by loosening byssal attachment resulting in loss of mussels due to wave action. Once a gap has been made in the bed, wave action, especially in winter, can enlarge the gap further. Similar effects have been reported to occur as a result of wave driven debris (e.g. logs) (see Suchanek & Seed, 1992). However, trampling adds an additional physical disturbance. Recovery in mussel beds is unpredictable, and may take several years and often longer in some environments (Suchanek & Seed, 1992).

3.3.6. Sabellariid worms

Sabellariid worms build tubes of concreted sand and large colonies can form raised biogenic reefs in the littoral zone (see Holt *et al.*, 1998). Zedler (1976, 1978; cited in Gharanshahi *et al.,* 1983) reported a decrease in abundance of the sabellariid worm *Phragmatophoma californica* in areas of heavy visitor use in California.

In the UK, littoral biogenic reefs are formed by *Sabellaria alveolata.* Cunningham *et al.* (1984) examined the effects of trampling on *Sabellaria alveolata* reefs. The reef recovered from the effects of trampling, (i.e. treading, walking or stamping on the reef structures) within 23 days. Recovery was achieved by repair of minor damage to the worm tube porches. However, severe damage from kicking and jumping on the reef structure, resulted in large cracks between the tubes, and removal of sections (ca 15x15x10 cm) of the structure (Cunningham *et al.*, 1984). Subsequent wave action enlarged the holes or cracks. However, after 23 days at one site, one side of the hole had begun to repair, and tubes had begun to extend into the eroded area. At another site, a smaller section (10x10x10 cm) was lost but after 23 days the space was already smaller due to rapid growth. Cunningham *et al.* (1984) reported that *Sabellaria alveolata* reefs were more tolerant of trampling than expected but noted that cracks could leave the reef susceptible to erosion and lead to large sections of the reef being washed away. However, eroded sections can survive and may lead to colonization of previously unsettled areas. The strange sculpturing of colonies in some areas is probably due to a combination of erosion and recovery (Cunningham *et al.*, 1984). Continuous trampling may be more detrimental and Holt *et al.* (1998) reported that, in Brittany, damage to reefs on popular beaches was limited to gaps created by trampling through the reef. Once gaps are formed, they may be enlarged by wave action as seen above.

In summary *Sabellaria alveolata* reefs are probably of intermediate intolerance to trampling and although worms can repair and stabilize the reefs relatively quickly, complete recovery will probably take several years once trampling has ceased. However, if a gap is formed, continuous trampling through the gap would probably remove any growing ‘crust’ of worms and the gap could not be repaired.

3.3.7. Shallow infralittoral algal communities

The effect of trampling on shallow algal communities was examined by a single Mediterranean study (Milazzo *et al.*, 2002). Experimental trampling of 18 transects were carried out at 0, 10, 25, 50, 100 and 150 passes and the community examined immediately after and three months later in the shallow infralittoral (0.3-0.5 m below mean low water). Percentage cover and canopy were significantly affected by trampling, the degree of effect increasing in proportion with trampling intensity. Intermediate trampling treatments (25, 50 and 100 tramples) were similar in effect but significantly different from 0 and 10 tramples. After 150 tramples, percentage cover was significantly lower. Erect macroalgae were particularly susceptible, e.g. the canopy forming *Cystoseira brachicarpa* v. *balearica* and *Dictyota mediterranea*. At low to intermediate trampling intensity, *Dictyota mediterranea* was strongly damaged while *Cystoseira brachicarpa* v. *balearica* lost fronds. At high trampling intensities, *D. mediterranea* was completely removed while *C. brachicarpa* v. *balearica* was reduced to holdfasts. Low to intermediate trampling intensities (10, 25, 50 tramples) resulted in a loss of algal biomass of 50 g/m^2, while 100 or 150 tramples resulted in a loss of ca 150 g/m^2. Recovery was incomplete after three months and significant differences in effect were still apparent between trampling treatments. Overall, trampling reduced percentage algal cover and canopy. However, the study focused on the canopy forming species and lower turf forming species were not mentioned.

In summary the above evidence suggests that shallow infralittoral algal communities are susceptible to the effects of trampling by pedestrians. Again the canopy forming, erect species seem to be the most susceptible. Trampling of sublittoral fringe communities could occur as coasteerers haul themselves out of the water at the bottom of the shore. Therefore, sublittoral fringe communities in the UK could be susceptible but there is no evidence at present.

3.3.8. Limpets and other gastropod grazers

Zedler (1976, 1978; cited in Gharanshahi *et al.,* 1983) reported that heavy visitor use was associated with lower abundances of the limpet *Collisella digitalis.* Jiggling of shells of *C. digitalis* to simulate an attempt at removal was sufficient to kill 12% of individuals, while removal of *C. digitalis* and *C. scabra* killed large numbers of individuals even when care was taken to replace them in their proper position. In the UK, large numbers of limpets (*Patella* spp.) kicked from rocks or removed in experimental studies die, presumably because the foot is damaged in the process, and/or they succumb to desiccation or predation (P. Moore, pers comm.). Gharanshahi *et al.* (1983) noted that *C. conus* increased in abundance, probably due to a decrease in competition from other limpet species.

In single step experiments, Povey & Keough (1991) noted that less than 5% of littorinids were crushed and overturned littorinids righted themselves before the next high tide. Kicking and stepping on the limpet *Cellana tramosirica* had little effect, with only four of 80 kicked limpets and only two of the 80 'stepped on' not being found the next day. Trampling experiments did not significantly affect the size distribution of the limpets *C. tramosirica* or *Austrocochlea constricta.* In coralline algal turf, the gastropod *Turbo undulatus* decreased in number in heavy trampling transect, while no significant change in number of the limpet *Patelloida alticostata* was found.

Limpet and non-limpet grazers were generally observed to increase in abundance in trampled brown algal mats, presumably due to removal of canopy forming brown algae that inhibit grazing and growth of the microalgae on which most gastropod grazers feed (Gharanshahi *et al.,* 1983; Povey & Keough, 1991; Keough & Quinn, 1998).

In summary trampling may crush a few individual limpets and gastropods but the majority will be unaffected. On algal dominated shores, removal of algal cover by trampling may benefit microalgal grazers. However, it should be noted that species that graze periphyton or macroalgae directly (e.g. *Littorina obtusata*, and some isopods) may be adversely affected. The major impact to limpets from recreational use is their removal by kicking or collecting as souvenirs.

3.3.9. Bare space

Several studies reported an increase in the abundance of bare space in response to trampling (Fletcher & Frid, 1996a, b; Schiel & Taylor, 1999; Jenkins *et al.*, 2002).

3.4. Factors influencing the effects of trampling

The above evidence demonstrates that the effects of trampling are dependant on the nature of the receiving community. However, the effect of trampling is also dependant on the following:

- trampling intensity (footsteps or tramples per unit area per unit time);
- trampling frequency (e.g. tramples per day, tramples per month);
- trampling duration (e.g. a single event, occasional or short term, long term or continuous), and
- the weight of the visitor and the footwear used.

The experimental studies detailed above demonstrated that the degree of trampling impact increased with increasing trampling intensity. Fletcher & Frid (1996a, b) demonstrated that continuous trampling significantly altered the algal community affected. However, Keough & Quinn (1998) reported that the effects of long term ‘pulses’ of trampling varied between sites, suggesting that the response to trampling is dependant on site specific variation in the community and its ability to recover.

Liddle (1997) noted that the impact from trampling was dependant on the force or pressure per unit area exerted by the visitor, which was in turn dependant on the weight of the visitor and the footwear used. For example, bare feet on hard ground produce a pressure of 297 g/cm^2, shoes 180 g/cm^2, and Vibram-soled boots (on hard ground) produce a pressure of 416 g/cm^2. In the studies above, footwear used in experimental trampling studies varied. For example, Povey & Keough (1991) used rubber-soled athletic shoes or sandals; Brosnan & Crumrine (1994) used rubber-soled shoes; and Schiel and Taylor (1999) used gumboots, while other studies did not specify. Curiously in South Africa, Bally & Griffiths (1989) found little difference in experimental trampling experiments, in which ‘neoprene thongs’ (flip-flops) were worn. Bally & Griffiths (1999) noted that 85% of visitors in their study area walked across the shore in bare feet, which forced the visitor to proceed with caution to prevent personal injury, and hence minimized damage.

4. Identification of vulnerable or at risk communities and species

The *MarLIN* approach to sensitivity assessment is similar to that used by recreational ecologists to categorise susceptible species (Liddle, 1991, 1997; see Appendix 2). Species are categorised by their resilience to damage (intolerance) and potential for subsequent recovery (recoverability), which in combination provides a category of ‘susceptibility’ (Liddle, 1991) or ‘sensitivity’ (Hiscock *et al.*, 1999; Tyler-Walters & Jackson, 1999; Tyler-Walters *et al.,* 2002, 2005).

Vulnerability assessment includes the concept of ‘exposure’ to an impact. Hence, a ‘vulnerable’ (or ‘at risk’) species or community is one that is exposed to an impact to which it is ‘sensitive’ (see Box 1). An approach to vulnerability or risk assessment of habitats important for commercial fisheries is outlined by Carlin & Rogers (2002), while a numerical approach is outlined by Oakwood Environmental (2002). However, both studies require estimates of the degree of exposure of sensitive communities and species to the impacting activity.

Green *et al.* (in prep) described the level of activity of coasteering at 13 sites in west Wales as high, medium or low but no definition of these terms was given. Therefore, no comparison with the experimental trampling data summarized above (section 3.2) could be made. Hence, no attempt has been made to rank sites by 'exposure' to trampling or coasteering nor to develop a scale of vulnerability within this report. Instead it has been assumed that sensitive communities or species that occur in the vicinity of sites at which coasteering occurs are 'potentially vulnerable' and that the level of vulnerability to trampling is dependant on the level of sensitivity to trampling and physical disturbance.

Biotopes that occur within a 0.5 km radius of sites subject to coasteering activity are listed in Appendix 1. The sensitivity of rocky intertidal and shallow infralittoral biotopes, within the vicinity of coasteering activities, researched by the *MarLIN* programme is shown in Appendix 4, together with a synopsis of the evidence used to assess sensitivity and the information used to derive their recoverabilities is given in Appendix 5.

Not all the rocky intertidal and infralittoral biotopes listed in Appendix 1 have been researched by the *MarLIN* programme. However, the sensitivity of researched biotopes has been used to 'represent' their sensitivity. A biotope was chosen as 'representative' of one or more other biotopes if the 'represented' biotope(s):

- occurred in similar habitats;
- was populated by similar functional groups of organisms, and
- was populated by the same (or functionally similar) species indicative of sensitivity as the biotope(s) they were chosen to represent.

The 'representative' biotopes have been researched as single entities. The 'representative' and their 'represented' biotope(s) are shown in Appendix 6. With the exception of LR.FLR.CvOV.FaCR, none of the 2004 (ver. 04.05) LR.FLR.CvOV biotopes in Appendix 1 have been researched, and no equivalents are known.

Please note sensitivity assessments and key information reviews are designed to provide the information required to make scientifically based environmental management decisions. It is not possible for sensitivity assessments to consider every possible outcome and are indicative. *MarLIN* sensitivity assessments are indicative qualitative judgements based on the best available scientific information. **They do not allow quantitative analysis.** The sensitivity assessments represent the most likely (or probable) result of a given change in an environmental factor on a species population or biotope. *MarLIN* sensitivity assessments should be read in conjunction with the explanation and key information provided.

The sensitivity assessment of rocky intertidal biotopes takes into account the effects of trampling as discussed in section 3.3 but also includes other sources of physical disturbance and addresses the impacts on the entire community. In addition, the *MarLIN* approach to sensitivity assessment is 'precautionary' in nature and where data is equivocal, the 'worst-case' scenario is recorded.

4.1. Potentially vulnerable rocky intertidal communities

The intolerance, recoverability and sensitivity of the researched biotopes, occurring in the vicinity of coasteering sites in west Wales are shown in Table 2. The barnacle dominated biotopes (e.g. ELR.Bpat), mussel dominated biotopes (e.g. MLR.MytB and MLR.MytFves) and coralline algal turf biotopes (e.g. ELR.Coff, and LR.Cor) have been assessed as of intermediate intolerance to trampling, i.e. their abundance and/or extent is likely to be decreased by trampling and physical disturbance, in agreement with the evidence above

(Section 3.3). However, their recovery potential is high, i.e. between one to five years, so that their overall sensitivity has been assessed as 'low'. Similarly, *Sabellaria alveolata* reefs were assessed as of low sensitivity to trampling, since they were able to recover within five years.

However, **recoverability assumes that physical disturbance has ceased**. Coasteering represents a seasonal event, and trampling has been shown (Section 3.3) to cause long term change, especially when trampling is continuous (Fletcher & Frid, 1996a, b; Keough & Quinn, 1998). Trampling from coasteering is likely to occur during the recovery phase of many of the biotopes shown in Table 2 and Appendix 4, since it occurs mainly in the spring and summer months, a period that coincides with the peak recruitment phase of many intertidal fucoids, red algae and barnacles. Therefore, intolerance (the communities' susceptibility to damage) is probably a better indicator of the potential vulnerability of rocky intertidal communities. The intolerance of likely vulnerable communities is emboldened in Table 2.

Nevertheless, a few exceptions stand out. Ephemeral green algal dominated biotopes (LR.G and MLR.Ent) are highly intolerant of physical disturbance but are dominated by opportunistic algae that recover quickly and may colonize disturbed habitats. Therefore, ephemeral green algal communities have been identified as potential vulnerable.

Fucoid dominated biotopes (e.g. MLR.BF and SLR.Asc) were assessed as highly intolerant of physical disturbance and trampling. Barnacles and most fucoids were considered to be able to recover within 5 years, and are of therefore assessed as of moderate sensitivity, suggesting a particular vulnerability to trampling. *Ascophyllum nodosum* is noted for poor recruitment and slow growth (Knight & Parke, 1950; Printz, 1959; Holt *et al.*, 1997) so that e.g. SLR.Asc is likely to be highly sensitive and hence particularly vulnerable to trampling.

Littoral overhang and cave biotopes (LR.Ov) were also assessed to be of high intolerance and hence moderate sensitivity, since these communities are composed of relatively delicate ascidians, sponges that are likely to be removed by physical disturbance and particularly abrasion. However, it should be noted that no information concerning the effects of trampling or visitor contact on overhang or cave communities was found.

4.2. Other potentially vulnerable communities

Supralittoral lichen (LR.YG), algal crust (LR.Chr) and shallow infralittoral and sublittoral fringe biotopes have been included in Table 2.

The yellow and grey lichen zone (LR.YG) may be particularly vulnerable to trampling. Fletcher (1980) noted that large specimens of lichens, e.g. *Ramalina siliquosa*, were only found on vertical rocks inaccessible to animals, including man. Trampling damage was greatest when the thallus was wet, causing it too peel from the surface, while when dry, some fragments were likely to remain to propagate the lichen (Fletcher, 1980). Physical disturbance of the lichen flora or substratum may reduce species richness and favour more rapid growing, disturbance tolerant species, e.g. *Lecanora dispersa*, *Candelariella vitellina* and *Rinodina gennerii* (Fletcher, 1980). Therefore, lichens were considered to be of intermediate intolerance of physical disturbance. However, growth rates are low, rarely more than 0.5-1 mm/year in crustose species while foliose species may grow up to 2-5 mm/year, and although ubiquitous, colonization is slow. Crump & Moore (1997) observed that lichens had not colonized experimentally cleared substrata within 12 months. Brown (1974) reported that recolonization of substrata within Caerthillian Cove, Cornwall, which was heavily affected by oil and dispersants after the *Torrey Canyon* oil spill, took 7 years to begin. Therefore, recoverability is likely to be low, and lichens may be highly sensitive of physical disturbance at the top of the shore.

Table 2. Potentially vulnerable rocky intertidal biotopes (or that represent biotopes) that occur in the vicinity of sites subject to coasteering in west Wales. The intolerance of potentially vulnerable communities is emboldened, and particularly vulnerable communities are shaded.

Biotope name	Biotope code	Intolerance	Recoverability	Sensitivity	Evidence / confidence
LITTORAL ROCK (and other hard substrata)					
LICHENS AND ALGAL CRUSTS					
Yellow and grey lichens on supralittoral rock.	**LR.L.YG**	**Intermediate**	**Low**	**High**	**Moderate**
Chrysophyceae on vertical upper littoral fringe soft rock.	**LR.L.Chr**	**Intermediate**	Very high	Low	Moderate
EXPOSED LITTORAL ROCK (mussel and barnacle shores)					
Mytilus edulis and barnacles on very exposed eulittoral rock.	**ELR.MB.MytB**	**Intermediate**	High	Low	Moderate
Barnacles and *Patella* spp. on exposed or moderately exposed, or vertical sheltered eulittoral rock.	**ELR.MB.Bpat**	**Intermediate**	High	Low	High
Corallina officinalis on very exposed lower eulittoral rock.	**ELR.FR.Coff**	**Intermediate**	Very high	Low	Moderate
Himanthalia elongata and red seaweeds on exposed lower eulittoral rock.	ELR.FR.Him	Low	High	Low	Moderate
MODERATELY EXPOSED LITTORAL ROCK (barnacle and fucoid shores)					
Barnacles and fucoids (moderately exposed shores).	**MLR.BF**	**High**	**High**	**Moderate**	Moderate
Fucus serratus and under-boulder fauna on lower eulittoral boulders.	**MLR.BF.Fser.Fser .Bo**	**High**	**High**	**Moderate**	Moderate
Enteromorpha spp. on freshwater influenced or unstable upper eulittoral rock.	**MLR.Eph.Ent**	High	Very high	Low	Moderate
Rhodothamniella floridula on sand-scoured lower eulittoral rock.	**MLR.Eph.Rho**	**Intermediate**	High	Low	Moderate
Mytilus edulis and *Fucus vesiculosus* on moderately exposed mid-eulittoral rock).	**MLR.MF.MytFves**	**Intermediate**	High	Low	Moderate
Sabellaria alveolata reefs on sand-abraded eulittoral rock.	**MLR.Sab.Salv**	**Intermediate**	High	Low	Moderate
SHELTERED LITTORAL ROCK (fucoid shores)					
Ascophyllum nodosum on very sheltered mid eulittoral rock.	**SLR.F.Asc**	**High**	**Low**	**High**	High
Fucus ceranoides on reduced salinity eulittoral rock.	**SLR.F.Fcer**	**Intermediate**	High	Low	Moderate

Biotope name	Biotope code	Intolerance	Recoverability	Sensitivity	Evidence / confidence
Barnacles and *Littorina littorea* on unstable eulittoral mixed substrata.	**SLR.FX.BLlit**	**Intermediate**	High	Low	Moderate
Fucus vesiculosus on mid eulittoral mixed substrata.	**SLR.FX.FvesX**	**Intermediate**	High	Low	Moderate
LITTORAL ROCK (other)					
Green seaweeds (*Enteromorpha* spp. and *Cladophora* spp.) in upper shore rockpools.	LR.Rkp.G	High	Very high	Low	Moderate
Corallina officinalis and coralline crusts in shallow eulittoral rockpools.	**LR.Rkp.Cor**	**Intermediate**	High	Low	Low
Fucoids and kelps in deep eulittoral rockpools.	**LR.Rkp.FK**	**Intermediate**	High	Low	Low
Seaweeds in sediment (sand or gravel)-floored eulittoral rockpools.	**LR.Rkp.SwSed**	**Intermediate**	High	Low	Low
Hydroids, ephemeral seaweeds and *Littorina littorea* in shallow eulittoral mixed substrata pools.	**LR.Rkp.H**	**Intermediate**	Very high	Low	Low
Overhangs and caves.	**LR.Ov**	**High**	**High**	**Moderate**	High
Rhodothamniella floridula in littoral fringe soft rock caves.	**LR.Ov.RhoCv**	**Intermediate**	High	Low	Moderate
Faunal crusts on littoral wave-surged cave walls.	**LR.FLR.CvOv .FaCr**[3]	**Intermediate**	High	Low	Moderate
INFRALITTORAL ROCK (and other hard substrata)					
EXPOSED INFRALITTORAL ROCK					
Alaria esculenta on exposed sublittoral fringe rock.	EIR.KfaR.Ala	Low	High	Low	Low
Laminaria hyperborea forest with a faunal cushion (sponges and polyclinids) and foliose red seaweeds on very exposed infralittoral rock.	**EIR.KfaR.LhypFa**	**Intermediate**	**Moderate**	**Moderate**	Moderate
Laminaria hyperborea with dense foliose red seaweeds on exposed infralittoral rock.	**EIR.KfaR.LhypR**	**Intermediate**	High	Low	Moderate
Foliose red seaweeds on exposed or moderately exposed lower infralittoral rock.	**EIR.KfaR.FoR**	**Intermediate**	High	Low	Low

Biotope name	Biotope code	Intolerance	Recoverability	Sensitivity	Evidence / confidence
Sponge crusts and anemones on wave-surged vertical infralittoral rock.	**EIR.SG.SCAn**	**High**	**High**	**Moderate**	High
MODERATELY EXPOSED INFRALITTORAL ROCK					
Laminaria digitata on moderately exposed sublittoral fringe rock.	**MIR.KR.Ldig.Ldig**	**Intermediate**	High	Low	Moderate
Laminaria digitata and piddocks on sublittoral fringe soft rock.	**MIR.KR.Ldig.Pid**	**Intermediate**	High	Low	Low
Laminaria saccharina, *Chorda filum* and dense red seaweeds on shallow unstable infralittoral boulders and cobbles.	**MIR.SedK.LsacChoR**	**Intermediate**	High	Low	Moderate
INFRALITTORAL ROCK (other)					
Alcyonium digitatum and a bryozoan, hydroid and ascidian turf on moderately exposed vertical infralittoral rock.	**IR.FaSwV.AlcByH**	**High**	**High**	**Moderate**	High

Shallow infralittoral and sublittoral fringe biotopes have been included to represent sublittoral fringe communities where coasteerers haul themselves out of the water, and communities that coasteerers may come into contact with on the vertical faces of gulley and plunge pools. Shallow water algal communities in the Mediterranean were shown to be susceptible to trampling (Milazzo *et al.*, 2002). Similar UK communities may show a similar intolerance, although no evidence was found. Pulling on kelps as hand holds while leaving the water may damage the stipe or growing meristem, resulting in loss of plants at that locality.

Please note, however, that the sensitivity assessments for the infralittoral biotopes are based on physical disturbance from anchorage, creels and similar activities, and are included to **highlight a potential vulnerability that requires further investigation**.

4.3. Potentially vulnerable important biotopes

The marine natural heritage importance of biotopes, identified within the vicinity of areas subject to coasteering is shown in Table 3 below. Potentially vulnerable biotopes are emboldened and particularly vulnerable communities are shaded.

Table 3 demonstrates that all the biotopes that occur within the vicinity of coasteering activities can be identified within Annex I habitats of the Habitats Directive (Brazier & Connor, 1999). Table 3 includes 13 nationally rare and scarce biotopes and 18 potentially vulnerable biotopes (and their representatives) that are listed as important in Wales.

4.4. Potentially vulnerable important species

The likely vulnerable species are detailed in Section 3.3, and include fucoids, erect articulated coralline algae, barnacles, and mussels. Species of conservation concern that are likely to occur within the rocky intertidal or sublittoral fringe on the Pembrokeshire coast (Moore, 2002) are listed in Table 4. Sensitivity information is only available for *Paludinella litorina*, which was assessed as of high sensitivity to physical disturbance on shingle shores. However, due to its small size and crevice habitat is would probably be protected from trampling on the rocky shore.

4.5. Potentially vulnerable areas of the coast

Tyler-Walters & Lear (2004) tagged Phase I biotope data and target notes, for the Pembrokeshire and Severn Estuary marine SACs with sensitivity to smothering, physical disturbance and hydrocarbon contamination. The sensitivity information was presented to CCW as a GIS layer, to overlay existing CCW Phase I survey data. The resultant GIS provides a tool to identify areas sensitive to, and hence potentially vulnerable to physical disturbance, including trampling.

However, as noted in Section 3.4.1 above, recoverability and sensitivity assume that the impact (e.g. trampling) ceases. Therefore where trampling is an annual activity, intolerance is probably a better indicator of potential vulnerability, and sensitivity may under-estimate impact. Nevertheless, the presence of moderately or higher sensitivity biotopes may indicate potentially vulnerable communities

Table 3. The marine natural heritage importance of researched (and their representative) biotopes in the vicinity of coasteering sites in west Wales. UK BAP = UK Biodiversity Action Plan. Reefs, caves, sand flats, sandbanks, bays, estuaries and lagoons refer to the relevant Annex I habitats of the Habitats Directive. Potentially vulnerable biotopes are emboldened and particularly vulnerable communities are shaded.

Habitat Name	Biotope code	Welsh Important	EC Habitats Directive	Reefs	Caves	Sand flats	Sandbanks	Bays	Estuaries	Lagoons	UK BAP	UK BAP Habitat	National Status
Yellow and grey lichens on supralittoral rock	**LR.YG**		●	●				●	●	●	●	Maritime cliff and slopes, Saline lagoons, Supralittoral rock (broad habitat statement) Littoral rock (broad habitat statement)	Widespread
Chrysophyceae on vertical upper littoral fringe soft rock	**LR.Chr**		●	●				●			●	Littoral and sublittoral chalk, Maritime cliff and slopes, Supralittoral rock (broad habitat statement)	Rare
***Mytilus edulis* and barnacles on very exposed eulittoral rock**	**ELR.MytB**		●	●				●			●	Littoral rock (broad habitat statement)	Common
Barnacles and *Patella* spp. on exposed or moderately exposed, or vertical sheltered, eulittoral rock	**ELR.BPat**		●	●				●	●	●	●	Saline lagoons, Littoral rock (broad habitat statement)	Widespread
***Corallina officinalis* on very exposed lower eulittoral rock**	**ELR.Coff**		●	●							●	Littoral rock (broad habitat statement)	Scarce
Himanthalia elongata and red seaweeds on exposed lower eulittoral rock	ELR.Him		●	●				●			●	Littoral rock (broad habitat statement)	Common
Barnacles and fucoids (moderately exposed shores)	**MLR.BF**			●				●	●		●	Littoral rock (broad habitat statement)	Widespread
Underboulder communities	**MLR.Fser.Fser.Bo**	●	●	●				●	●		●	Littoral rock (broad habitat statement)	Common

Habitat Name	Biotope code	Welsh Important	EC Habitats Directive	Reefs	Caves	Sand flats	Sandbanks	Bays	Estuaries	Lagoons	UK BAP	UK BAP Habitat	National Status
Enteromorpha spp. on freshwater-influenced or unstable upper eulittoral rock	MLR.Ent		●	●				●	●	●	●	Saline lagoons, Littoral rock (broad habitat statement)	Uncommon
***Rhodothamniella floridula* on sand-scoured lower eulittoral rock**	**MLR.Rho**		●	●				●	●	●	●	Saline lagoons, Littoral rock (broad habitat statement)	Uncommon
***Mytilus edulis* and *Fucus vesiculosus* on moderately exposed mid eulittoral rock**	**MLR.MytFves**		●	●				●	●		●	Littoral rock (broad habitat statement)	Scarce
***Sabellaria alveolata* reefs on sand-abraded eulittoral rock**	**MLR.Salv**	●	●	●				●	●		●	*Sabellaria alveolata* reefs, Littoral rock (broad habitat statement)	Scarce
***Ascophyllum nodosum* on very sheltered mid eulittoral rock.**	**SLR.Asc**		●	●				●	●	●	●	Saline lagoons, Littoral rock (broad habitat statement)	Widespread
***Fucus ceranoides* on reduced salinity eulittoral rock**	**SLR.Fcer**		●	●					●	●	●	Saline lagoons, Littoral rock (broad habitat statement)	Scarce
Barnacles and *Littorina littorea* on unstable eulittoral mixed substrata	**SLR.BLlit**		●	●				●	●	●	●	Saline lagoons, Littoral rock (broad habitat statement)	Rare
***Fucus vesiculosus* on mid eulittoral mixed substrata**	**SLR.FvesX**		●	●				●	●	●	●	Saline lagoons, Littoral rock (broad habitat statement)	Common
Green seaweeds (*Enteromorpha* spp. and *Cladophora* spp.) in upper shore rockpools	LR.G		●	●				●	●		●	Littoral rock (broad habitat statement), Supralittoral rock (broad habitat statement)	Widespread

Habitat Name	Biotope code	Welsh Important	EC Habitats Directive	Reefs	Caves	Sand flats	Sandbanks	Bays	Estuaries	Lagoons	UK BAP	UK BAP Habitat	National Status
***Corallina officinalis* and coralline crusts in shallow eulittoral rockpools** **Represents**	**LR.Cor**	●	●	●				●	●	●	●	Saline lagoons, Littoral rock (broad habitat statement)	Widespread
Coralline crusts and *Paracentrotus lividus* in shallow eulittoral rockpools.	LR.Rkp.Cor.Par	●		●				●	●	●	●	Littoral rock (broad habitat statement)	Rare
Bifurcaria birfurcata in shallow eulittoral rockpools.	LR.Rkp.Cor.Bif	●		●				●	●	●	●	Littoral rock (broad habitat statement)	Rare
Cystoseira spp. in shallow eulittoral rockpools.	LR.Rkp.Cor.Cys	●		●				●	●	●	●	Littoral rock (broad habitat statement)	Rare
Fucoids and kelps in deep eulittoral rockpools	**LR.FK**	●	●	●				●	●				Common
Seaweeds in sediment (sand or gravel)-floored eulittoral rockpools	**LR.SwSed**	●	●	●				●	●				Common
Hydroids, ephemeral seaweeds and *Littorina littorea* in shallow eulittoral mixed substrata pools	**LR.H**	●	●	●				●	●				Rare
Overhangs and caves **Represents**	**LR.Ov**		●	●	●			●	●		●	Littoral rock (broad habitat statement)	Scarce
Sponges and shade tolerant red seaweeds on overhanging lower shore bedrock.	LR.Ov.SR[1]	●	●	●	●			●	●		●	Littoral rock (broad habitat statement)	Common
Sponges, bryozoans and ascidians on deeply overhanging lower shore bedrock.	LR.Ov.SByAs*2*	●	●	●	●			●	●		●	Littoral rock (broad habitat statement)	Uncommon
***Rhodothamniella floridula* in upper littoral fringe soft rock caves**	**LR.RhoCv**		●	●	●			●	●		●	Littoral and sublittoral chalk Maritime cliff and slopes Littoral rock (broad habitat statement)	Rare

[1] Equivalent to biotope (version 04.05) 'Sponges and shade-tolerant red seaweeds on overhanging lower eulittoral bedrock and in cave entrances' (LR.FLR.CvOv.SpR) (Connor *et al.,* 2004).

2 Equivalent to biotope (version 04.05) 'Sponges, bryozoans and ascidians on deeply overhanging lower shore bedrock or caves' (LR.FLR.CvOv.SpByAs) (Connor *et al.,* 2004).

Habitat Name	Biotope code	Welsh Important	EC Habitats Directive	Reefs	Caves	Sand flats	Sandbanks	Bays	Estuaries	Lagoons	UK BAP	UK BAP Habitat	National Status
Faunal crusts on littoral wave-surged cave walls	**LR.FLR.CvOv .FaCr**	●	?	?	?	?	?	?	?	?			Not available
Alaria esculenta on exposed sublittoral fringe bedrock	EIR.Ala		●	●							●	Inshore sublittoral rock (broad habitat statement)	Widespread
Laminaria hyperborea **forest with a faunal cushion (sponges and polyclinids) and foliose red seaweeds on very exposed upper infralittoral rock**	**EIR.LhypFa**		●	●				●			●	Inshore sublittoral rock (broad habitat statement)	Uncommon
Laminaria hyperborea **with dense foliose red seaweeds on exposed infralittoral rock.**	**EIR.LhypR**		●	●				●			●	Inshore sublittoral rock (broad habitat statement)	Not available
Foliose red seaweeds on exposed or moderately exposed lower infralittoral rock	**EIR.FoR**		●	●				●			●	Inshore sublittoral rock (broad habitat statement)	Not available
Sponge crusts and anemones on wave-surged vertical infralittoral rock. **Represents**	**EIR.SCAn**	●	●	●	●			●			●	Inshore sublittoral rock (broad habitat statement)	Scarce
Sponge crusts and anemones and *Tubularia indivisa* in shallow infralittoral surge gullies.	EIR.SG.SCAn .Tub	●	●	●	●			●			●	Inshore sublittoral rock (broad habitat statement)	Not available
Sponge crusts and ascidians on wave-surged vertical infralittoral rock.	EIR.SG.SCAs	●	●	●	●			●			●	Inshore sublittoral rock (broad habitat statement)	Not available
Dendrodoa grossularia and *Clathrina coriacea* on wave-surged vertical infralittoral rock.	EIR.SG.SCAs .DenCla	●	●	●	●			●			●	Inshore sublittoral rock (broad habitat statement)	Scarce
Sponge crusts, colonial (polyclinid) ascidians and a bryozoan/hydrozoan turf on wave-surged vertical or overhanging infralittoral rock.	EIR.SG.SCAs .ByH	●	●	●	●			●			●	Inshore sublittoral rock (broad habitat statement)	Not available
Sponge crusts on extremely wave-surged infralittoral cave or gulley walls.	EIR.SG.SC	●	●	●	●			●			●	Inshore sublittoral rock (broad habitat statement)	Not available
Laminaria digitata **on moderately exposed sublittoral fringe rock**	**MIR.Ldig.Ldig**		●	●				●	●	●	●	Saline lagoons, Inshore sublittoral rock (broad habitat statement)	Widespread

Habitat Name	Biotope code	Welsh Important	EC Habitats Directive	Reefs	Caves	Sand flats	Sandbanks	Bays	Estuaries	Lagoons	UK BAP	UK BAP Habitat	National Status
Laminaria digitata **and piddocks on sublittoral fringe soft rock**	**MIR.Ldig.Pid**	●	●	●				●	●		●	Littoral and sublittoral chalk	Scarce
Laminaria saccharina*, *Chorda filum **and dense red seaweeds on shallow unstable infralittoral boulders or cobbles**	**MIR.LsacChoR**		●	●							●	Sublittoral sands and gravels, Inshore sublittoral rock (broad habitat statement)	Not available
Alcyonium digitatum **with a bryozoan, hydroid and ascidian turf on moderately exposed vertical infralittoral rock.**	**IR.AlcByH**		●	●				●			●	Littoral and sublittoral chalk Inshore sublittoral rock (broad habitat statement)	Common

Table 4. Welsh species of conservation concern that occur within the rocky intertidal or sublittoral fringe of Pembrokeshire marine SAC (Moore, 2002).

Species name	Common name	Environmental position	National status
Anthopleura thallia	Glaucus pimplet	Lower eulittoral & sublittoral fringe	Scare
Aiptasia mutabilis	Trumpet anemone	Lower eulittoral to sublittoral	Scarce
Balanophyllia regia	Scarlet and Gold Star Coral	Lower eulittoral to upper infralittoral	Scarce
Allomelita pellucida	Amphipod	Intertidal or shallow subtidal	Rare or scarce
Dromia personata	Sponge crab	All sublittoral zones and occasionally lower shore	Scarce
Paludinella litorina	Snail	Supralittoral and littoral	Scarce
Gelidiella calcicola	Red alga	Infralittoral from extreme low water to 30m	Rare
Gigartina pistillata	Red alga	Lower eulittoral	Scarce
Pterosiphonia pennata	Red alga	Lower eulittoral to infralittoral	Scarce
Zanardinia prototypes	Brown alga	Upper infralittoral	Scarce
Padina pavonica	Peacock's Tail	Lower eulittoral	Scarce

5. Discussion

The studies reviewed in Section 3.3 demonstrated that trampling represents an additional and unique impact on rocky shore communities. The level of impact is dependent on the shore type, the community present and the intensity, frequency and duration of the trampling, as well as the footwear used.

The literature suggests that the brown algal shrub communities and their understorey are probably the most sensitive communities. On wave sheltered to moderately exposed shores trampling by visitors has been implicated in a general decline in foliose (canopy forming) algal cover, an increase in algal turf species in some cases, and an increase in bare space and microalgal grazing gastropods (e.g. limpets). Collection of souvenirs has been implicated in a decrease in limpet abundance. On barnacle dominated shores, barnacle cover may decline, and erect coralline algal turf may also be impacted, although the results vary between studies.

The *MarLIN* approach to sensitivity assessment has enabled potentially vulnerable biotopes that occur in the vicinity of sites subject to coasteering, to be identified. However, the term 'potentially vulnerable' must be emphasised, due to the nature of coasteering and any resultant trampling.

The evidence reviewed in Section 3.3 demonstrated that trampling effects were highly localized, especially at high intensities. High intensity or continuous trampling is associated with the formation of clearly visible, and in some cases bare, paths through the environment in terrestrial systems, e.g. wildlife reserves and sand dunes (Liddle, 1997). Similar paths have been observed through brown algal shrubs (Brosnan, 1993; Fletcher & Frid, 1996a, b) and sabellariid reefs (Holt *et al.*, 1998) in the marine environment. However, the effects may vary between sites (see Keough & Quinn, 1998)

Similarly, coasteering activities are likely to follow regular ‘routes’ through the rocky intertidal, dictated by the conditions, the accessibility of rock, cliffs, or caves, and the desire to find positions from which coasteerers can jump into the water or gulley below. As a result, the routes chosen, and especially areas where coasteerers gather prior to ‘jumping’, are likely to receive high intensities of trampling. Therefore, the evidence suggests that coasteering could result in detectable paths through rocky intertidal communities, depending on the frequency of the activity at any one site.

Coasteering is of concern because it provides access to hitherto inaccessible, and possibly pristine, marine habitats. Coasteering occurs on relatively wave exposed coasts, with steep rocky surfaces to jump from, and plunge-pools and gulleys to experience. Wave exposed coasts are likely to be dominated by barnacle communities in the intertidal, with mixed barnacles and mussels on sloping surfaces, and coralline turfs in more wave exposed areas. Rockpools, caves, overhangs, vertical rocks and gullies provide additional interest. However, the majority of the evidence available has been obtained on moderately wave exposed or wave sheltered shores, with sloping rock platforms that are relatively accessible to the general public. In addition, the majority of the studies were carried out overseas, where the climate and species (and their recovery characteristics) differ from those found in the UK.

Nevertheless, the potentially vulnerable biotopes include Welsh important biotopes, together with nationally rare and scarce biotopes, e.g. ELR.Coff (exposed erect coralline turfs), rare coralline rock pools, and cave or overhang communities. Cave, overhangs, gullies and rockpool habitats may also provide habitats for rare and scarce species.

Due to the localized nature of the activity it is not possible to predict the exact communities, and hence biotopes, that coasteering activity will encounter. Therefore, while biotopes that occur in the vicinity of coasteering activities may be intolerant of trampling, there is presently no information on the exact routes used and hence no information concerning the intolerant biotopes impacted. Hence, the biotopes identified above can only be regarded as of ‘potential vulnerability’.

The effect of trampling on wave exposed rocky intertidal communities, especially caves and gullies, and sublittoral fringe communities requires further study before the impact of coasteering within Pembrokeshire marine SAC can be assessed. But coasteering activities themselves provide the perfect opportunity to study trampling on wave exposed rocky intertidal and sublittoral fringe communities.

Brosnan (1993), Brosnan & Crumrine (1994) and Fletcher & Frid (1996a, b) identified three management strategies:

1. closure or removal of access;
2. sacrificial areas, or
3. rotation of areas.

Closure or removal of access is difficult to enforce and unpopular. While recreational activities in the marine environment may have adverse impacts, engaging the public with the marine environment has the potential to increase understanding and the perceived ‘value’ of the environment to the public. The public, and hence the country, are more likely to protect areas of the environment that they value, so that closure may be detrimental to management in the long term.

Areas that are already used, or impacted may be left as ‘sacrificial’ areas. Therefore, a network of designated paths or routes that avoid particularly rare, scarce or vulnerable communities but do not adversely affect the ‘experience’ of coasteering may be beneficial. An alternative is to ‘rotate’ routes, so that the communities are undisturbed for a period of time that allows the resident community to recover, depending on its recoverability.

Strategies two and three will both benefit from educational material to raise the public's awareness of the environment they are visiting.

Therefore, proactive management may minimize the impact of coasteering. Voluntary codes of practice for similar outdoor pursuits such as climbing and gorge-walking exist. For example, the British Mountaineering Club (BMC) publishes a code of practice and guidance on restricted climbing periods to minimize disturbance to nesting birds, has developed a good practice booklet, 'Tread Lightly', aimed at hill walkers and mountaineers, and is actively involved in environmental conservation. Similarly, 'Sustainable Use of Snowdonia' publishes a booklet to promote the sustainable use of the Afon Ddu gorge.

Coasteering groups could be involved in developing a conservation policy that includes gains for our knowledge of impacts and the occurrence of species that are indicators of change (e.g. climate change species or non-native species).

6. Conclusions

1. The impacts of trampling on rocky shore communities are relatively poorly studied, especially in the UK.
2. Most of the available literature focuses on sloping, publicly accessible shores that are moderately wave exposed or sheltered and, therefore, may not be directly applicable to the types of shore on which coasteering activities occur.
3. The available evidence suggests that foliose canopy forming algae (e.g. fucoids) are particularly intolerant and sensitive to trampling impacts. Trampling was shown to damage erect coralline turfs barnacles, and result in an increase in bare space. In some cases paths across the shore were visible.
4. On brown algae dominated shores, understorey algae may suffer due to increased desiccation but algal turf species, opportunists and gastropod grazers (e.g. limpets) may increase in abundance as an indirect effect of trampling.
5. Trampling impacts result from physical contact and wear and are dependant on the intensity, duration, and frequency of trampling, and even the type of footwear used.
6. *MarLIN* sensitivity assessment can be used to identify potentially vulnerable biotopes but where coasteering is a seasonal activity, recovery may not occur. Therefore, intolerance is a better indicator of potential vulnerability.
7. A total of 19 intolerant rocky intertidal biotopes were identified as potentially vulnerable to trampling and hence coasteering within the Pembrokeshire marine SAC, of which six are of Welsh importance and eight are nationally rare or scarce.
8. In addition, attention has been drawn to the potential vulnerability of delicate slow growing supralittoral lichen communities and sublittoral fringe communities.
9. Nevertheless, trampling is a highly localized impact and is has not been possible to identify biotopes, and hence communities, actually impacted by coasteering activities in the Pembrokeshire marine SAC.
10. Direct survey of the routes used by coasteering groups within the Pembrokeshire marine SAC is required to identify the intensity, duration and frequency of trampling impact, together with the communities impacted.
11. Given the paucity of data concerning trampling effects in the rocky intertidal in the UK, a survey of the impacts of coasteering would provide an opportunity to examine the effects of trampling and visitor use on steep rocky, wave exposed shores.

7. Recommendations

1. The routes used by coasteering groups in the Pembrokeshire marine SAC should be surveyed directly, to establish the trampling intensity and the communities present and hence any intolerant and/or important biotopes and species affected.
2. Coasteering groups should avoid rare and scarce species or biotopes if present and where possible.
3. Any management should be proactive, and engage with the coasteering proprietors to develop a management regime. Wherever possible, past experience and environmental codes of practice developed by other outdoor organizations should be drawn on as models for good environmental conduct and management practice.
4. A ‘minimizing your footprint’ guide for coasteerers should be produced and made available to all coasteering groups.
5. Coasteering groups should be encouraged to ‘put something back’ by undertaking recording designed for volunteers to assess climate change impacts and non-native species concerns.

8. References

(* = unseen by author)

Amsler, C.D. & Searles, R.B., 1980. Vertical distribution of seaweed spores in a water column off shore of North Carolina. *Journal of Phycology*, **16**, 617-619.

Austin, A.P., 1960a. Life history and reproduction of *Furcellaria fastigiata* (L.) Lamouroux. *Annals of Botany, New Series*, **24**, 257-274.

Austin, A.P., 1960b. Observations on the growth, fruiting and longevity of *Furcellaria fastigiata* (L.) Lamouroux. *Hydrobiologia*, **15**, 193-207.

Bally, R. & Griffiths, C.L., 1989. Effects of human trampling on an exposed rocky shore. *International Journal of Environmental Studies*, **34**, 115-125.

Bamber, R.N. & Irving, P.W., 1993. The *Corallina* run-offs of Bridgewater Bay. *Porcupine Newsletter*, **5**, 190-197.

Brazier, D.P. & Connor, D.W. (2nd edn.), 1999. *Relationship between Annex I marine habitats of the EC Habitats Directive and the MNCR BioMar marine biotope classification*. Joint Nature Conservation Committee, Peterborough. [JNCC Marine Information Notes, no. 8].

Beauchamp, K.A. & Gowing, M.M., 1982. A quantitative assessment of human trampling effects on a rocky intertidal community. *Marine Environmental Research*, **7**, 279-293.

Bennell, S.J., 1981. Some observations on the littoral barnacle populations of North Wales. *Marine Environmental Research*, **5**, 227-240.

Bird, C.J., Saunders, G.W. & McLachlan, J., 1991. Biology of *Furcellaria lumbricalis* (Hudson) Lamouroux (Rhodophyta: Gigartinales), a commercial carrageenophyte. *Journal of Applied Phycology*, **3**, 61-82.

Birkett, D.A., Maggs, C.A., Dring, M.J. & Boaden, P.J.S., 1998b. Infralittoral reef biotopes with kelp species: an overview of dynamic and sensitivity characteristics for conservation management of marine SACs. *Natura 2000 report prepared by Scottish Association of Marine Science (SAMS) for the UK Marine SACs Project.*

Boalch, G.T. & Jephson, N.A., 1981. A re-examination of the seaweeds on Colman's traverses at Wembury. *Proceedings of the International Seaweed Symposium*, **8**, 290-293.

Boalch, G.T., Holme, N.A., Jephson, N.A. & Sidwell, J.M.C., 1974. A resurvey of Colman's intertidal traverses at Wembury, South Devon. *Journal of the Marine Biological Association of the United Kingdom*, **5**, 551-553.

Bradshaw, C., Veale, L.O., Hill, A.S. & Brand, A.R., 2000. The effects of scallop dredging on gravelly seabed communities. In: *Effects of fishing on non-target species and habitats* (ed. M.J. Kaiser & de S.J. Groot), pp. 83-104. Oxford: Blackwell Science.

Brault, S. & Bourget, E., 1985. Structural changes in an estuarine subtidal epibenthic community: biotic and physical causes. *Marine Ecology Progress Series*, **21**, 63-

Brosnan, 1993. The effect of human trampling on biodiversity of rocky shores: monitoring and management strategies. *Recent Advances in Marine Science and Technology '92*, 333-341.

*Brosnan, D.M. & Crumrine, L.L., 1992. Human impact and a management strategy for Yaquina Head Outstanding Natural Area. *A report to the Bureau of Land Management, Department of the Interior, Salem, Oregon.*

Brosnan, D.M. & Crumrine, L.L., 1994. Effects of human trampling on marine rocky shore communities. *Journal of Experimental Marine Biology and Ecology*, **177**, 79-97.

*Brosnan, D.M., Elliot, J., Quon, I., 1996. The effect of trampling on marine rocky shores in southern California. *California Sea Grant. Biennial Report Of Completed Projects* 1992-94, pp. 172-174.

Brown, P.J. & Taylor, R.B., 1999. Effects of trampling by humans on animals inhabiting coralline algal turf in the rocky intertidal. *Journal of Experimental Marine Biology and Ecology*, **235**, 45-53.

Bucklin, A., 1985. Biochemical genetic variation, growth and regeneration of the sea anemone, *Metridium*, of British shores. *Journal of the Marine Biological Association of the United Kingdom*, **65**, 141-157.

Carlin, D. & Rogers, S., 2002. A procedure to assess the effects of dredging on commercial fisheries. *Report to Department for the Environment, Food and Rural Affairs from the Centre for Environment, Fisheries, and Aquaculture Science (CEFAS) Lowestoft.*

Castric, A., 1977. Recrutement et succession du benthos rocheux sublittoral. In *Proceedings of the Eleventh European Symposium on Marine Biology, University College, Galway, 5-11 October 1976. Biology of benthic organisms* (ed. B.F. Keegan *et al*.), pp 147-154. Oxford: Pergamon Press

Castric-Fey, A., 1983. Recruitment, growth and longevity of *Pomatoceros triqueter* and *Pomatoceros lamarckii* (Polychaeta, Serpulidae) on experimental panels in the Concarneau area, South Brittany. *Annales de l'Institut Oceanographique, Paris*, **59**, 69-91.

Chamberlain, Y.M., 1996. Lithophylloid Corallinaceae (Rhodophycota) of the genera *Lithophyllum* and *Titausderma* from southern Africa. *Phycologia*, **35**, 204-221.

Chia, F.S. & Spaulding, J.G., 1972. Development and juvenile growth of the sea anemone *Tealia crassicornis*. *Biological Bulletin, Marine Biological Laboratory, Woods Hole,* **142**, 206-218.

Christie, H., Fredriksen, S. & Rinde, E., 1998. Regrowth of kelp and colonization of epiphyte and fauna community after kelp trawling at the coast of Norway. *Hydrobiologia*, **375/376**, 49-58.

Colman, J., 1933. The nature of the intertidal zonation of plants and animals. *Journal of the Marine Biological Association of the United Kingdom*, **18**, 435-476.

Connor, D.W., Brazier, D.P., Hill, T.O., & Northen, K.O., 1997a. Marine Nature Conservation Review: marine biotope classification for Britain and Ireland. Volume 1. Littoral biotopes. Version 97.06. *Joint Nature Conservation Committee, Peterborough, JNCC Report*, no. 229.

Connor, D.W., Dalkin, M.J., Hill, T.O., Holt, R.H.F. & Sanderson, W.G., 1997b. Marine Nature Conservation Review: marine biotope classification for Britain and Ireland. Volume 2. Sub-littoral biotopes. Version 97.06. *Joint Nature Conservation Committee, Peterborough*, *JNCC Report*, no. 230.

Connor, D.W., Allen, J.H., Golding, N., Lieberknecht, L.M., Northen, K.O. & Reker, J.B., 2003. *The National Marine Habitat Classification for Britain and Ireland Version 03.02.* [On-line] Peterborough: Joint Nature Conservation Committee [ISBN 1 86107 546 4] Available from <http://www.jncc.gov.uk/marine/biotopes/default.htm>

Cornelius, P.F.S., 1992. Medusa loss in leptolid Hydrozoa (Cnidaria), hydroid rafting, and abbreviated life-cycles among their remote island faunae: an interim review. *Scientia Maritima*, **56**, 245-261. [*Proceedings of 2nd International Workshop of the Hydrozoan Society, Spain, September 1991. Aspects of hydrozoan biology* (ed. Bouillon, J., F. Cicognia, J.M. Gili & R.G. Hughes).]

Cornelius, P.F.S., 1995b. *North-west European thecate hydroids and their medusae. Part 2. Sertulariidae to Campanulariidae.* Synopses of the British Fauna (New Series) (ed. R.S.K. Barnes & J.H. Crothers), The Linnean Society of London. Shrewsbury: Field Studies Council. [Synopses of the British Fauna no. 50]

Crump, R. & Moore, J., 1997. Monitoring of upper littoral lichens at Sawdern Point. *Report to the Shoreline and Terrestrial Task Group, Sea Empress Environmental Evaluation Committee (SEEEC), July 1997*.

Cullinane, J.P., McCarthy, P. & Fletcher, A., 1975. The effect of oil pollution in Bantry Bay. *Marine Pollution Bulletin*, **6**, 173-176.

Daly, M.A. & Mathieson, A.C., 1977. The effects of sand movement on intertidal seaweeds and selected invertebrates at Bound Rock, New Hampshire, USA. *Marine Biology*, **43**, 45-55.

Denis, T.G. & Murray, S.N., 2001. Among-site variation in the effects of trampling disturbance on *Silvetia compressa* (O. Fucales) populations. *Journal of Phycology*, **37 (s3)**, 16 (abstract only).

Dethier, M.N., 1984. Disturbance and recovery in intertidal pools: maintenance of mosaic patterns. *Ecological Monographs*, **54**, 99-118.

Dons, C., 1927. Om Vest og voskmåte hos *Pomatoceros triqueter*. *Nyt Magazin for Naturvidenskaberne*, **LXV**, 111-126.

Eno, N.C., Clark, R.A. & Sanderson, W.G. (ed.), 1997. *Non-native marine species in British waters: a review and directory.* Peterborough: Joint Nature Conservation Committee.

Erickson, A., Klinger, T. & Fradkin, S.C., 2004. A pilot study of the effects of human trampling on rocky intertidal areas in Olympic National Park, USA [On-line]. In *Proceedings of the Georgia Basin/Puget Sound Research Conference, Vancouver, BC (Canada), 31 Mar-3 Apr 2003* (ed. T.W. Droscher & D.A. Fraser) 5 pp. Available from <http://www.psat.wa.gov/Publications/03_proceedings/start.htm>

Fletcher, R.L. & Callow, M.E., 1992. The settlement, attachment and establishment of marine algal spores. *British Phycological Journal*, **27**, 303-329.

Fletcher, A., 1980. Marine and maritime lichens of rocky shores: their ecology, physiology and biological interactions. In *The Shore Environment, vol. 2: Ecosystems* (ed. J.H. Price, D.E.G. Irvine & W.F. Farnham), pp. 789-842. London: Academic Press. [Systematics Association Special Volume no. 17(b)].

Fletcher, H. & Frid, C.L.J., 1996a. The response of an intertidal algal community to persistent trampling and the implications for rocky shore management. In *Studies in European Coastal Management* (ed. P.S. Jones, M.G. Healy & A.T. Williams), pp. 233-245. Cardigan: Samara Publishing

Fletcher, H. & Frid, C.L.J., 1996b. Impact and management of visitor pressure on rocky intertidal algal communities. *Aquatic Conservation: Marine & Freshwater Ecosystems*, **6**, 287-297.

Ghazanshahi, J., Huchel, T.D. & Devinny, J.S., 1983. Alteration of southern California rocky shore ecosystems by public recreational use. *Journal of Environmental Management*, **16**, 379-394.

Gili, J-M. & Hughes, R.G., 1995. The ecology of marine benthic hydroids. *Oceanography and Marine Biology: an Annual Review*, **33**, 351-426.

Green, S., Bowles, J. & G. King, in prep. West Wales Coastal Recreation Audit – Phase One Report (2nd draft). *Report to the Pembrokeshire Coastal Forum and Partners by Bowles Green Limited, Oswaldkirk.*

Gubbay, S., 1983. Compressive and adhesive strengths of a variety of British barnacles. *Journal of the Marine Biological Association of the United Kingdom*, **63**, 541-555.

Harlin, M.M., & Lindbergh, J.M., 1977. Selection of substrata by seaweed: optimal surface relief. *Marine Biology*, **40**, 33-40.

Hartnoll, R.G., 1975. The annual cycle of *Alcyonium digitatum*. *Estuarine and Coastal Marine Science*, **3**, 71-78.

Hartnoll, R.G., 1998. Circalittoral faunal turf biotopes: An overview of dynamics and sensitivity characteristics for conservation management of marine SACs. *Scottish Association of Marine Sciences, Oban, Scotland.* [UK Marine SAC Project. Natura 2000 reports.]

Hartnoll, R.G. & Hawkins, S.J., 1985. Patchiness and fluctuations on moderately exposed rocky shores. *Ophelia*, **24**, 53-63.

Hatcher, A.M., 1998. Epibenthic colonization patterns on slabs of stabilised coal-waste in Poole Bay, UK. *Hydrobiologia*, **367**, 153-162.

Hawkins, S.J. & Harkin, E., 1985. Preliminary canopy removal experiments in algal dominated communities low on the shore and in the shallow subtidal on the Isle of Man. *Botanica Marina*, **28**, 223-30.

Hawkins, S.J. & Hartnoll, R.G., 1985. Factors determining the upper limits of intertidal canopy-forming algae. *Marine Ecology Progress Series*, **20**, 265-271.

Hawkins, S.J. & Southward, A.J., 1992. The *Torrey Canyon* oil spill: recovery of rocky shore communities. In *Restoring the Nations Marine Environment*, (ed. G.W. Thorpe), Chapter 13, pp. 583-631. Maryland, USA: Maryland Sea Grant College.

Hayward, P.J. & Ryland, J.S. (ed.), 1995b. *Handbook of the marine fauna of North-West Europe.* Oxford: Oxford University Press.

Hill, S., Burrows, S.J. & Hawkins, S.J., 1998. *Intertidal Reef Biotopes (Volume VI). An overview of dynamics and sensitivity characteristics for conservation management of marine Special Areas of Conservation.* Oban: Scottish Association for Marine Science (UK Marine SACs Project).

Hiscock, K., Jackson, A. & Lear, D., 1999. Assessing seabed species and ecosystem sensitivities: existing approaches and development, October 1999 edition. *Report to the Department of Environment, Transport and the Regions from the Marine Life Information Network (MarLIN).* Plymouth: Marine Biological Association of the UK.

van den Hoek, C., 1982. The distribution of benthic marine algae in relation to the temperature regulation of their life histories. *Biological Journal of the Linnean Society*, **18**, 81-144.

Holt, T.J., Jones, D.R., Hawkins, S.J. & Hartnoll, R.G., 1995. The sensitivity of marine communities to man induced change - a scoping report. *Countryside Council for Wales, Bangor, Contract Science Report,* no. 65.

Holt, T.J., Hartnoll, R.G. & Hawkins, S.J., 1997. The sensitivity and vulnerability to man-induced change of selected communities: intertidal brown algal shrubs, *Zostera* beds and *Sabellaria spinulosa* reefs. *English Nature, Peterborough, English Nature Research Report* No. 234.

Holt, T.J., Rees, E.I., Hawkins, S.J. & Seed, R., 1998. *Biogenic reefs (Volume IX). An overview of dynamic and sensitivity characteristics for conservation management of marine SACs.* Scottish Association for Marine Science (UK Marine SACs Project), 174 pp.

Irvine, L. M. & Chamberlain, Y. M., 1994. *Seaweeds of the British Isles,* vol. 1. *Rhodophyta,* Part 2B *Corallinales, Hildenbrandiales.* London: Her Majesty's Stationery Office.

James, P.W. & Syratt, W.J., 1987. Air quality monitoring at Sullom Voe using lichens as biological indicator species: results of the 1986 repeat photographic survey. *Report to the Shetland Oil Terminal Environmental Advisory Group (SOTEAG) (Monitoring Sub Group), February 1987.*

Jenkins, C., Haas, M.E., Olson, A. & Ruesink, J.L., 2002. Impacts of trampling on a rocky shoreline of San Juan Island, Washington, USA. *Natural Areas Journal*, **22**, 260-269.

Jensen, A.C., Collins, K.J., Lockwood, A.P.M., Mallinson, J.J. & Turnpenny, W.H., 1994. Colonization and fishery potential of a coal-ash artificial reef, Poole Bay, United Kingdom. *Bulletin of Marine Science*, **55**, 1263-1276.

Kain, J.M., 1975. Algal recolonization of some cleared subtidal areas. *Journal of Ecology*, **63**, 739-765.

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

Kenny, A.J. & Rees, H.L., 1994. The effects of marine gravel extraction on the macrobenthos: early post dredging recolonisation. *Marine Pollution Bulletin*, **28**, 442-447.

Keough, M.J. & Quinn, G.P., 1991. Causality and the choice of measurements for detecting human impacts in marine environments. *Australian Journal of Marine and Freshwater Research*, **42**, 539-554.

Keough, M.J. & Quinn, G.P., 1998. Effects of periodic disturbances from trampling on rocky intertidal algal beds. *Ecological Applications*, **8**, 141-161.

Kitching, J.A., 1937. Studies in sublittoral ecology. II. Recolonization at the upper margin of the sublittoral region; with a note on the denudation of *Laminaria* forest by storms. *Journal of Ecology*, **25**, 482-495.

Kitching, J.A. & Thain, V.M., 1983. The ecological impact of the sea urchin *Paracentrotus lividus* (Lamarck) in Lough Ine, Ireland. *Philosophical Transactions of the Royal Society of London, Series B*, **300**, 513-552.

Knight, M. & Parke, M., 1950. A biological study of *Fucus vesiculosus* L. and *Fucus serratus* L. *Journal of the Marine Biological Association of the United Kingdom*, **29**, 439-514.

Laffoley, D.A., Connor, D.W., Tasker, M.L. & Bines, T., 2000. *Nationally important seascapes, habitats and species. A recommended approach to their identification, conservation and protection*, pp. 17. Peterborough: English Nature.

Liddle, M.J., 1991. Recreational ecology: effects of trampling on plants and corals. *Trends in Ecology and Evolution*, **6**, 13-17.

Liddle, M.J., 1997. *Recreational ecology. The ecological impact of outdoor recreation and ecotourism.* London: Chapman & Hall.

Littler, M.M., & Kauker, B.J., 1984. Heterotrichy and survival strategies in the red alga Corallina officinalis L. *Botanica Marina*, **27**, 37-44.

Lutz, R.A. & Kennish, M.J., 1992. Ecology and morphology of larval and early larval postlarval mussels. In *The mussel Mytilus: ecology, physiology, genetics and culture,* (ed. E.M. Gosling), pp. 53-85. Amsterdam: Elsevier Science Publ. [Developments in Aquaculture and Fisheries Science, no. 25]

MacFarlane, C.I., 1952. A survey of certain seaweeds of commercial importance in southwest Nova Scotia. *Canadian Journal of Botany*, **30**, 78-97.

Mathieson, A.C. & Burns, R.L., 1975. Ecological studies of economic red algae. 5. Growth and reproduction of natural and harvested populations of *Chondrus crispus* Stackhouse in New Hampshire. *Journal of Experimental Marine Biology and Ecology*, **17**, 137-156.

Menot, L., Chassé, C. & Kerambrun, L., 1998. Experimental study of the ecological impact of hot water / high pressure cleaning on rocky shores. *Proceedings of the 21st Arctic and Marine Oil Spill Program (AMOP) technical seminar, Edmonton, Alberta, Canada, June 10-12 1998,* vol. 2, pp. 891-901.

Metaxas, A. & Scheibling, R.E., 1993. Community structure and organization of tidepools. *Marine Ecology Progress Series*, **98**, 187-198.

Milazzo, M., Chemello, R., Badalamenti, F. & Riggio, S., 2002. Short-term effect of human trampling on the upper infralittoral macroalgae of Ustica Island MPA (western Mediterranean, Italy). *Journal of the Marine Biological Association of the United Kingdom,* **82**, 745-748.

Minchinton, T.E., Schiebling, R.E. & Hunt, H.L., 1997. Recovery of an intertidal assemblage following a rare occurrence of scouring by sea ice in Nova Scotia, Canada. *Botanica Marina*, **40**, 139-148.

Moore, J., 2002. An atlas of marine Biodiversity Action Plan species and habitats and Species of Conservation Concern in Wales, 2nd edn. *Report to Cyngor Cefn Gwlad Cymru / Countryside Council for Wales, CCW Contract Science Report* no. 509.

Murray, S.N., Goodson, J., Gerrard, A. & Luas, T., 2001. Long-term changes in rocky intertidal seaweed populations in urban southern California. *Journal of Phycology*, **37 (s3)**, 37 (abstract only).

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

Oakwood Environmental, 2002. Development of a methodology for the assessment of cumulative effects of marine activities using Liverpool Bay as a case study. *Report from Oakwood Environmental Ltd to the Countryside Council for Wales, CCW Contract Science Report*, no. 522.

Paine, R.T. & Levin, S.A., 1981. Intertidal landscapes: disturbance and the dynamics of pattern. *Ecological Monographs*, **51**, 145-178.

Povey, A. & Keough, J., 1991. Effects of trampling on plant and animal populations on rocky shores. *Oikos*, **61**, 355-368.

Printz, H.S., 1959. Investigations of the failure of recuperation and re-populating in cropped *Ascophyllum* areas. *Avhandlinger utgitt av Det Norske Videnskap-Akademi i Oslo* No. 3.

Raffaelli, D. & Hawkins, S., 1999. *Intertidal Ecology* 2nd edn.. London: Kluwer Academic Publishers.

Schiel, D.R. & Taylor, D.I., 1999. Effects of trampling on a rocky intertidal algal assemblage in southern New Zealand. *Journal of Experimental Marine Biology and Ecology*, **235**, 213-235.

Sebens, K.P., 1985. Community ecology of vertical rock walls in the Gulf of Maine: small-scale processes and alternative community states. In *The Ecology of Rocky Coasts: essays presented to J.R. Lewis, D.Sc.,* (ed. P.G. Moore & R. Seed), pp. 346-371. London: Hodder & Stoughton Ltd.

Sebens, K.P., 1986. Spatial relationships among encrusting marine organisms in the New England subtidal zone. *Ecological Monographs*, **56**, 73-96.

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

Smith, J.E. (ed.), 1968. 'Torrey Canyon'. *Pollution and marine life.* A report by the Plymouth Laboratory of the Marine Biological Association of the United Kingdom. Cambridge: Cambridge University Press.

Solé-Cava, A.M., Thorpe, J.P., & Todd, C.D., 1994. High genetic similarity between geographically distant populations in a sea anemone with low dispersal capabilities. *Journal of the Marine Biological Association of the United Kingdom*, **74**, 895-902.

Sousa, W.P., Schroeter, S.C. & Daines, S.D., 1981. Latitudinal variation in intertidal algal community structure: the influence of grazing and vegetative propagation. *Oecologia*, **48**, 297-307.

Southward, A.J. & Southward, E.C., 1978. Recolonisation of rocky shores in Cornwall after use of toxic dispersants to clean up the *Torrey Canyon* spill. *Journal of the Fisheries Research Board of Canada*, **35**, 682-706.

South West Watersports, 2004. *The official guide to watersports. Coasteering* Exeter: Marine Southwest Taskforce. Available from <http:// www.sw-watersports.com>.

Sutherland, J.P. & Karlson, R.H., 1977. Development and stability of the fouling community at Beaufort, North Carolina. *Ecological Monographs*, **47**, 425-446.

Stegenga, H., 1978. The life histories of *Rhodochorton purpureum* and *Rhodochorton floridulum* (Rhodophyta, Nemiales) in culture. *British Phycological Journal*, **13**, 279-289.

Stengel, D.B., Wilkes, R.J. & Guiry, M.D., 1999. Seasonal growth and recruitment of *Himanthalia elongata* (Fucales, Phaeophycota) in different habitats on the Irish west coasts. *European Journal of Phycology*, **34**, 213-221.

Todd, C.D., 1994. Competition for space in encrusting bryozoan assemblages: the influence of encounter angle, site and year. *Journal of the Marine Biological Association of the United Kingdom*, **74**, 603-622.

Tyler-Walters, H. & Hiscock, K., 2003. A biotope sensitivity database to underpin delivery of the Habitats Directive and Biodiversity Action Plan in the seas around England and Scotland. *Report to English Nature and Scottish Natural Heritage from the Marine Life Information Network (MarLIN).* Plymouth: Marine Biological Association of the UK. [English Nature Research Reports, ENRR No. 499.]

Tyler-Walters, H. & Jackson, A., 1999. Assessing seabed species and ecosystems sensitivities. Rationale and user guide, June 2000 edition. *Report to the Department of the Environment Transport and the Regions from the Marine Life Information Network (MarLIN).* Plymouth: Marine Biological Association of the UK.

Tyler-Walters, H. & Lear, D.B., 2004. Sensitivity mapping for Oil Pollution Incident Response. *Report to Cyngor Cefn Gwlad Cymru / Countryside Council for Wales from the Marine Life Information Network (MarLIN).* Plymouth: Marine Biological Association of the UK. [Contract no. FC 73-02-282]

Tyler-Walters, H., Ager, O.E.D. & Hiscock, K., 2002. Development of a marine sensitivity mapping database and GIS integration. Stage 1. Review of current habitat and species information. *Report to Cyngor Cefn Gwlad Cymru / Countryside Council for Wales from the Marine Life Information Network (MarLIN).* Plymouth: Marine Biological Association of the UK. [Contract no. FC 73-02-245]

Tyler-Walters, H., Hiscock, K., Lear, D.B. & Jackson, A., 2001. Identifying species and ecosystem sensitivities. *Report to the Department for Environment, Food and Rural Affairs from the Marine Life Information Network (MarLIN).* Plymouth: Marine Biological Association of the UK. [Contract CW0826].

Tyler-Walters, H., Marshall, C., Hiscock, K., Hill, J.M., Budd, G.C.,Rayment, W.J. & Jackson, A., 2005. Description, temporal variation, sensitivity and monitoring of important marine biotopes in Wales. *Report to Cyngor Cefn Gwlad Cymru / Countryside Council for Wales from the Marine Life Information Network (MarLIN).* Plymouth: Marine Biological Association of the UK. [CCW Contract no. FC 73-023-255G]

Veale, L.O., Hill, A.S., Hawkins, S.J. & Brand, A.R., 2000. Effects of long term physical disturbance by scallop fishing on subtidal epifaunal assemblages and habitats. *Marine Biology*, **137**, 325-337.

Wernberg, T., Thomsen, M.S., Staehr, P.A. & Pedersen, M.F., 2001. Comparative phenology of *Sargassum muticum* and *Halidrys siliquosa* (Phaeophyceae: Fucales) in Limfjorden, Denmark. *Botanica Marina*, **44**, 31-39.

Wilson, D.P., 1929. The larvae of the British sabellarians. *Journal of the Marine Biological Association of the United Kingdom,* **16**, 221-269.

*Zedler, J.B., 1976. *Ecological resource inventory of the Cabrillo National Monument Intertidal Zone.* 1976 Project Report. San Diego, California: San Diego State University Biology Department.

*Zedler, J.B., 1978. *Public use effects in the Cabrillo National Monument Intertidal Zone.* Project Report. San Diego, California: San Diego State University Biology Department.

Appendix 1. List of phase I biotopes within 0.5 km radius of sites subject to Coasteering activities in west Wales.

Aber-Eiddi

Biotope	Lifeform	Area (ha)	Notes
EIR.KFaR.Ala.Ldig	Kelp	0.0519	
EIR.KFaR.Ala.Ldig	Kelp	0.0513	
EIR.KFaR.Ala.Ldig	Kelp	0.4538	
EIR.SG.SCAs.DenCla	Short faunal turf; crusts & cushions	0.0075	
ELR.FR.Coff	Fucoids	0.1879	
ELR.FR.Coff	Fucoids	0.0316	
ELR.FR.Coff	Fucoids	0.0816	
ELR.FR.Him	Fucoids	0.0567	
ELR.FR.Him	Fucoids	0.0215	
ELR.FR.Him	Fucoids	0.0387	
ELR.FR.Him	Fucoids	0.0958	
ELR.MB.BPat	Mussels & Barnacles	0.0066	
ELR.MB.BPat	Mussels & Barnacles	0.4947	
ELR.MB.BPat	Mussels & Barnacles	0.0406	
ELR.MB.BPat	Mussels & Barnacles	0.1478	
ELR.MB.BPat	Mussels & Barnacles	0.0377	
ELR.MB.BPat	Mussels & Barnacles	0.0320	
ELR.MB.BPat	Mussels & Barnacles	0.0080	
ELR.MB.BPat	Mussels & Barnacles	0.0853	
ELR.MB.BPat	Mussels & Barnacles	0.7593	
ELR.MB.BPat	Mussels & Barnacles	0.6730	
ELR.MB.BPat	Mussels & Barnacles	0.0135	
ELR.MB.BPat	Mussels & Barnacles	0.0101	
ELR.MB.BPat	Mussels & Barnacles	0.0039	
ELR.MB.BPat	Mussels & Barnacles	0.0594	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0420	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0115	
LGS.S.AEur	Sand	1.2794	
LGS.S.AEur	Sand	0.0527	
LGS.Sh.BarSh	Shingle, coarse sand, mixed sediments	0.0141	
LGS.Sh.BarSh	Shingle, coarse sand, mixed sediments	0.5306	
LGS.Sh.BarSh	Shingle, coarse sand, mixed sediments	0.0499	
LGS.Sh.BarSh	Shingle, coarse sand, mixed sediments	0.0750	
LGS.Sh.BarSh	Shingle, coarse sand, mixed sediments	0.0167	
LGS.Sh.BarSh	Shingle, coarse sand, mixed sediments	0.0432	
LR.L.Ver.B	Lichens & algae	0.1569	
LR.L.Ver.Ver	Lichens & algae	0.1197	
LR.L.Ver.Ver	Lichens & algae	0.0169	
LR.L.Ver.Ver	Lichens & algae	0.0030	
LR.L.Ver.Ver	Lichens & algae	0.1371	
LR.L.Ver.Ver	Lichens & algae	0.0016	
LR.L.Ver.Ver	Lichens & algae	0.0007	
LR.L.Ver.Ver	Lichens & algae	0.0260	
LR.L.Ver.Ver	Lichens & algae	0.0000	
LR.L.YG	Lichens & algae	0.3080	

Biotope	Lifeform	Area (ha)	Notes
LR.L.YG	Lichens & algae	0.0674	
LR.L.YG	Lichens & algae	0.0130	
LR.L.YG	Lichens & algae	0.0006	
LR.Rkp.Cor	Algal turf	0.0025	Target Note 14
LR.Rkp.Cor.Bif	Algal turf	0.0025	Target Note 25
LR.Rkp.FK	Fucoids	0.0025	Target Note 24
LR.Rkp.G	Algal turf	0.0025	Target Note 23
LR.Rkp.G	Algal turf	0.0025	Target Note 18
MIR.KR.Ldig.Ldig	Kelp	0.0519	
MIR.KR.Ldig.Ldig	Kelp	0.0180	
MIR.KR.Ldig.Ldig	Kelp	0.0385	
MIR.KR.Ldig.Ldig	Kelp	0.0700	inferred biotope
MIR.KR.Ldig.Ldig	Kelp	0.1846	inferred biotope
MIR.SedK.Sac	Kelp	0.0615	
MLR.BF.Fser.Fser.Bo	Fucoids	0.0325	
MLR.BF.Fser.R	Fucoids	0.0102	
MLR.BF.Fser.R	Fucoids	0.0942	
MLR.BF.Fser.R	Fucoids	0.0486	
MLR.BF.Fser.R	Fucoids	0.1431	
MLR.BF.Fser.R	Fucoids	0.1680	
MLR.Eph.EntPor	Algal turf	0.0170	
MLR.Eph.EntPor	Algal turf	0.0144	
MLR.Eph.EntPor	Algal turf	0.0472	
MLR.Eph.EntPor	Algal turf	0.0395	
MLR.R.Osm	Algal turf	0.0049	
MLR.R.Osm	Algal turf	0.0032	
MLR.R.Osm	Algal turf	0.0029	
MLR.R.Osm	Algal turf	0.0041	
MLR.R.Osm	Algal turf	0.0364	
MLR.R.Osm	Algal turf	0.1620	
SLR.F.Fspi	Fucoids	0.0371	
SLR.F.Fspi	Fucoids	0.0353	
SLR.F.Fspi	Fucoids	0.2062	
SLR.F.Pel	Fucoids	0.0025	Target Note 17
SLR.F.Pel	Fucoids	0.0059	
SLR.FX.AscX	Fucoids	0.0046	
SLR.FX.AscX	Fucoids	0.0064	

Porth Clais

Biotope	Lifeform	Area (ha)	notes
EIR.KFaR.Ala.Ldig	Kelp	0.0062	
EIR.KFaR.Ala.Ldig	Kelp	0.0088	
EIR.KFaR.Ala.Ldig	Kelp	0.3287	
EIR.KFaR.Ala.Ldig	Kelp	0.1801	
EIR.KFaR.Ala.Ldig	Kelp	0.0162	
ELR.FR.Him	Fucoids	0.2735	
ELR.MB.BPat	Mussels & Barnacles	0.0064	
ELR.MB.BPat	Mussels & Barnacles	3.3695	
ELR.MB.BPat.Cat	Mussels & Barnacles	0.0025	Target Note 11
ELR.MB.BPat.Fvesl	Mussels & Barnacles	0.0096	
ELR.MB.BPat.Fvesl	Mussels & Barnacles	0.2180	
ELR.MB.BPat.Fvesl	Mussels & Barnacles	0.3819	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0056	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0062	

Biotope	Lifeform	Area (ha)	notes
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0045	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0031	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0120	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0607	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0608	
LGS.S.AEur	Sand	0.1069	
LGS.Sh.BarSh	Shingle, coarse sand, mixed sediments	0.2993	
LGS.Sh.BarSh	Shingle, coarse sand, mixed sediments	0.0343	
LMU.SMu.HedMac.Are	Mud	0.0524	
LR.L.Ver.B	Lichens & algae	0.0077	
LR.L.Ver.B	Lichens & algae	0.0044	
LR.L.Ver.B	Lichens & algae	0.0059	
LR.L.Ver.B	Lichens & algae	0.0036	
LR.L.Ver.B	Lichens & algae	0.0016	
LR.L.Ver.B	Lichens & algae	0.0111	
LR.L.Ver.B	Lichens & algae	0.0750	
LR.L.Ver.B	Lichens & algae	0.0733	
LR.L.Ver.Por	Lichens & algae	0.0025	Target Note 15
LR.L.Ver.Ver	Lichens & algae	0.0041	
LR.L.Ver.Ver	Lichens & algae	0.0041	
LR.L.Ver.Ver	Lichens & algae	0.0076	
LR.L.Ver.Ver	Lichens & algae	0.0055	
LR.L.Ver.Ver	Lichens & algae	0.0021	
LR.L.Ver.Ver	Lichens & algae	0.0014	
LR.L.Ver.Ver	Lichens & algae	0.0093	
LR.L.Ver.Ver	Lichens & algae	0.0069	
LR.L.Ver.Ver	Lichens & algae	0.0385	
LR.L.Ver.Ver	Lichens & algae	0.3833	
LR.L.Ver.Ver	Lichens & algae	0.1027	
LR.L.Ver.Ver	Lichens & algae	0.0815	
LR.L.YG	Lichens & algae	0.0035	
LR.L.YG	Lichens & algae	0.0065	
LR.L.YG	Lichens & algae	0.0974	
LR.L.YG	Lichens & algae	0.0046	
LR.L.YG	Lichens & algae	0.0018	
LR.L.YG	Lichens & algae	0.0009	
LR.L.YG	Lichens & algae	0.0080	
LR.L.YG	Lichens & algae	0.0299	
LR.L.YG	Lichens & algae	0.0132	
LR.L.YG	Lichens & algae	0.0055	
LR.L.YG	Lichens & algae	0.0425	
LR.L.YG	Lichens & algae	0.3855	
LR.L.YG	Lichens & algae	0.0826	
MLR.BF.PelB	Fucoids	0.0041	
MLR.Eph.Ent	Algal turf	0.0611	
MLR.Eph.Ent	Algal turf	0.0382	
MLR.R.XR	Algal turf	0.2964	
MLR.R.XR	Algal turf	0.0147	
MLR.R.XR	Algal turf	0.1185	
MLR.R.XR	Algal turf	0.1687	
SLR.F.Asc.Asc	Fucoids	0.0387	
SLR.F.Asc.Asc	Fucoids	0.0275	
SLR.F.Fspi	Fucoids	0.0626	

Biotope	Lifeform	Area (ha)	notes
SLR.F.Fspi	Fucoids	0.0043	
SLR.F.Fspi	Fucoids	0.0301	
SLR.F.Pel	Fucoids	0.0218	
SLR.F.Pel	Fucoids	0.0622	
SLR.FX.FcerX	Fucoids	0.1603	
SLR.FX.FcerX	Fucoids	0.4397	

Caerfai Bay

Biotope	Lifeform	Area (ha)	notes
EIR.KFaR.Ala.Ldig	Kelp	0.0062	
EIR.KFaR.Ala.Ldig	Kelp	0.0088	
EIR.KFaR.Ala.Ldig	Kelp	0.3287	
EIR.KFaR.Ala.Ldig	Kelp	0.1801	
EIR.KFaR.Ala.Ldig	Kelp	0.0162	
ELR.FR.Him	Fucoids	0.2735	
ELR.MB.BPat	Mussels & Barnacles	0.0064	
ELR.MB.BPat	Mussels & Barnacles	3.3695	
ELR.MB.BPat.Cat	Mussels & Barnacles	0.0025	Target Note 11
ELR.MB.BPat.Fvesl	Mussels & Barnacles	0.0096	
ELR.MB.BPat.Fvesl	Mussels & Barnacles	0.2180	
ELR.MB.BPat.Fvesl	Mussels & Barnacles	0.3819	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0233	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0211	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0056	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0062	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0045	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0031	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0120	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0607	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0608	
LGS.S.AEur	Sand	0.1069	
LGS.Sh.BarSh	Shingle, coarse sand, mixed sediments	0.2993	
LGS.Sh.BarSh	Shingle, coarse sand, mixed sediments	0.2688	
LGS.Sh.BarSh	Shingle, coarse sand, mixed sediments	0.0343	
LMU.SMu.HedMac.Are	Mud	0.0524	
LR.L.Ver.B	Lichens & algae	0.0077	
LR.L.Ver.B	Lichens & algae	0.0216	
LR.L.Ver.B	Lichens & algae	0.0197	
LR.L.Ver.B	Lichens & algae	0.0044	
LR.L.Ver.B	Lichens & algae	0.0059	
LR.L.Ver.B	Lichens & algae	0.0036	
LR.L.Ver.B	Lichens & algae	0.0016	
LR.L.Ver.B	Lichens & algae	0.0111	
LR.L.Ver.B	Lichens & algae	0.0750	
LR.L.Ver.B	Lichens & algae	0.0733	
LR.L.Ver.Por	Lichens & algae	0.0025	Target Note 15
LR.L.Ver.Ver	Lichens & algae	0.0041	
LR.L.Ver.Ver	Lichens & algae	0.0185	
LR.L.Ver.Ver	Lichens & algae	0.0179	
LR.L.Ver.Ver	Lichens & algae	0.0041	
LR.L.Ver.Ver	Lichens & algae	0.0076	

Biotope	Lifeform	Area (ha)	notes
LR.L.Ver.Ver	Lichens & algae	0.0055	
LR.L.Ver.Ver	Lichens & algae	0.0021	
LR.L.Ver.Ver	Lichens & algae	0.0014	
LR.L.Ver.Ver	Lichens & algae	0.0093	
LR.L.Ver.Ver	Lichens & algae	0.0069	
LR.L.Ver.Ver	Lichens & algae	0.0385	
LR.L.Ver.Ver	Lichens & algae	0.1027	
LR.L.Ver.Ver	Lichens & algae	0.0815	
LR.L.YG	Lichens & algae	0.0169	
LR.L.YG	Lichens & algae	0.0169	
LR.L.YG	Lichens & algae	0.0166	
LR.L.YG	Lichens & algae	0.0035	
LR.L.YG	Lichens & algae	0.0065	
LR.L.YG	Lichens & algae	0.0974	
LR.L.YG	Lichens & algae	0.0046	
LR.L.YG	Lichens & algae	0.0018	
LR.L.YG	Lichens & algae	0.0009	
LR.L.YG	Lichens & algae	0.0080	
LR.L.YG	Lichens & algae	0.0299	
LR.L.YG	Lichens & algae	0.0132	
LR.L.YG	Lichens & algae	0.0055	
LR.L.YG	Lichens & algae	0.0425	
LR.L.YG	Lichens & algae	0.0826	
MLR.BF.PelB	Fucoids	0.0041	
MLR.Eph.Ent	Algal turf	0.0611	
MLR.Eph.Ent	Algal turf	0.0382	
MLR.R.XR	Algal turf	0.2964	
MLR.R.XR	Algal turf	0.0147	
MLR.R.XR	Algal turf	0.1185	
MLR.R.XR	Algal turf	0.1687	
SLR.F.Asc.Asc	Fucoids	0.0387	
SLR.F.Asc.Asc	Fucoids	0.0275	
SLR.F.Fspi	Fucoids	0.0626	
SLR.F.Fspi	Fucoids	0.0043	
SLR.F.Fspi	Fucoids	0.0301	
SLR.F.Pel	Fucoids	0.0218	
SLR.F.Pel	Fucoids	0.0622	
SLR.FX.FcerX	Fucoids	0.1603	
SLR.FX.FcerX	Fucoids	0.4397	

Madoc's Haven, North Haven & Druidstron Haven.

Biotope	Lifeform	Area (ha)	notes
EIR.KFaR.LhypFa	Kelp	0.0304	
EIR.KFaR.LhypFa	Kelp	0.0514	
EIR.KFaR.LhypFa	Kelp	0.2032	
EIR.KFaR.LhypFa	Kelp	0.1404	
EIR.SG.SCAs	Short faunal turf; crusts & cushions	0.0047	
EIR.SG.SCAs	Short faunal turf; crusts & cushions	0.0041	
EIR.SG.SCAs	Short faunal turf; crusts & cushions	0.0025	Target Note 11
ELR.FR.Him	Fucoids	0.0025	Target Note 10
ELR.MB.BPat	Mussels & Barnacles	0.2021	
ELR.MB.BPat	Mussels & Barnacles	0.0128	
ELR.MB.BPat	Mussels & Barnacles	0.1501	
ELR.MB.BPat	Mussels & Barnacles	0.0516	

Biotope	Lifeform	Area (ha)	notes
ELR.MB.BPat.Fvesl	Mussels & Barnacles	0.1394	
ELR.MB.BPat.Fvesl	Mussels & Barnacles	0.0652	
ELR.MB.BPat.Fvesl	Mussels & Barnacles	0.1458	
ELR.MB.BPat.Fvesl	Mussels & Barnacles	0.1383	
ELR.MB.BPat.Fvesl	Mussels & Barnacles	0.1346	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0461	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0243	
ELR.MB.BPat.Sem	Mussels & Barnacles	1.1130	
ELR.MB.MytB	Mussels & Barnacles	0.0028	
ELR.MB.MytB	Mussels & Barnacles	0.0029	
ELR.MB.MytB	Mussels & Barnacles	0.0024	
ELR.MB.MytB	Mussels & Barnacles	0.0026	
ELR.MB.MytB	Mussels & Barnacles	0.0025	
ELR.MB.MytB	Mussels & Barnacles	0.1264	
ELR.MB.MytB	Mussels & Barnacles	1.0125	
ELR.MB.MytB	Mussels & Barnacles	0.0030	
ELR.MB.MytB	Mussels & Barnacles	0.0021	
ELR.MB.MytB	Mussels & Barnacles	0.0374	
ELR.MB.MytB	Mussels & Barnacles	0.0159	
ELR.MB.MytB	Mussels & Barnacles	0.0042	
LGS.Sh.BarSh	Shingle, coarse sand, mixed sediments	0.8415	
LGS.Sh.BarSh	Shingle, coarse sand, mixed sediments	0.3072	
LGS.Sh.BarSh	Shingle, coarse sand, mixed sediments	0.0788	
LGS.Sh.BarSh	Shingle, coarse sand, mixed sediments	0.4827	
LR.CvOv.SpR.Ov	Short faunal turf; crusts & cushions	0.0025	Target Note 9
LR.L.Ver.Ver	Lichens & algae	0.9609	
LR.L.YG	Lichens & algae	1.0422	
LR.Rkp.Cor.Bif	Algal turf	0.0025	Target Note 13
LR.Rkp.FK	Fucoids	0.0025	Target Note 13
LR.Rkp.G	Algal turf	0.0025	Target Note 9
MIR.KR.Ldig.Ldig	Kelp	0.0251	
MIR.KR.Ldig.Ldig	Kelp	0.1064	
MIR.KR.Ldig.Ldig	Kelp	0.2773	
MIR.KR.Ldig.Ldig	Kelp	0.1039	
MIR.SedK.Sac	Kelp	0.0230	
MLR.BF.Fser.R	Fucoids	0.0800	
MLR.BF.FvesB	Fucoids	0.0640	
MLR.Eph.Ent	Algal turf	0.0025	Target Note 6
MLR.Eph.EntPor	Algal turf	0.0735	
MLR.Eph.EntPor	Algal turf	0.1364	
MLR.MF.MytFR	Mussels & Barnacles	0.1180	

Castlemartin

Biotope	Lifeform	Area (ha)	notes
LGS.S.AP.P	Sand	0.7483	
LGS.S.Tal	Sand	0.2053	
LR.Rkp.G	Algal turf	0.0025	Target Note 9
MLR.Eph.Ent	Algal turf	0.5584	
MLR.Eph.EntPor	Algal turf	0.3841	
ELR.MB.BPat.Cat	Mussels & Barnacles	0.0025	Target Note 2

Biotope	Lifeform	Area (ha)	notes
ELR.MB.BPat.Cat	Mussels & Barnacles	0.3680	
ELR.MB.BPat.Cht	Mussels & Barnacles	0.1057	
ELR.MB.BPat.Cht	Mussels & Barnacles	0.2523	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.9629	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.6671	
ELR.MB.BPat.Sem	Mussels & Barnacles	2.5859	
ELR.MB.BPat.Sem	Mussels & Barnacles	0.7614	
ELR.MB.BPat.Sem	Mussels & Barnacles	1.5291	
LGS.S.AEur	Sand	0.0670	
LGS.S.AEur	Sand	1.3829	
LGS.S.Tal	Sand	0.6481	
LGS.Sh.BarSh	Shingle, coarse sand, mixed sediments	0.3227	
LR.CvOv.SpR.Ov	Short faunal turf; crusts & cushions	0.0025	Target Note 18
LR.L.Ver.Ver	Lichens & algae	0.4247	
LR.L.Ver.Ver	Lichens & algae	0.6888	
LR.L.Ver.Ver	Lichens & algae	0.4560	
LR.L.YG	Lichens & algae	0.3582	
LR.L.YG	Lichens & algae	0.5083	
LR.L.YG	Lichens & algae	0.1424	
LR.Rkp.Cor	Algal turf	0.0025	Target Note 5
LR.Rkp.Cor	Algal turf	0.0025	Target Note 6
LR.Rkp.Cor	Algal turf	0.0025	Target Note 6
LR.Rkp.Cor	Algal turf	0.0025	Target Note 3
LR.Rkp.Cor.Bif	Algal turf	0.0025	Target Note 6
LR.Rkp.Cor.Bif	Algal turf	0.0025	Target Note 6
LR.Rkp.Cor.Bif	Algal turf	0.0025	Target Note 4
LR.Rkp.FK	Fucoids	0.0025	Target Note 6
LR.Rkp.FK	Fucoids	0.0025	Target Note 6
LR.Rkp.FK	Fucoids	0.0060	
LR.Rkp.FK	Fucoids	0.0048	
LR.Rkp.G	Algal turf	0.0025	Target Note 5
LR.Rkp.SwSed	Fucoids	0.0025	Target Note 6
LR.Rkp.SwSed	Fucoids	0.0025	Target Note 6
LR.Rkp.SwSed	Fucoids	0.0025	Target Note 14
MIR.KR.Ldig.Ldig	Kelp	0.6935	
MIR.KR.Ldig.Ldig	Kelp	0.7791	
MIR.KR.Ldig.Ldig	Kelp	1.2809	
MLR.BF.Fser.Fser	Fucoids	0.0246	
MLR.BF.Fser.Fser	Fucoids	0.2957	
MLR.BF.Fser.Fser	Fucoids	0.0305	
MLR.BF.Fser.R	Fucoids	0.3568	
MLR.BF.FvesB	Fucoids	0.4731	
MLR.BF.FvesB	Fucoids	0.2328	
MLR.BF.FvesB	Fucoids	0.0388	
MLR.BF.PelB	Fucoids	0.0105	
MLR.BF.PelB	Fucoids	0.0184	
MLR.Eph.Ent	Algal turf	0.0342	
MLR.R.Mas	Algal turf	0.0025	Target Note 12
MLR.R.Mas	Algal turf	0.2363	
MLR.R.Mas	Algal turf	0.4751	
MLR.R.Mas	Algal turf	0.0808	
MLR.R.Mas	Algal turf	0.0647	
MLR.R.Osm	Algal turf	0.3908	
MLR.R.Osm	Algal turf	0.0631	

Biotope	Lifeform	Area (ha)	notes
MLR.R.XR	Algal turf	0.1627	
SLR.F.Asc.Asc	Fucoids	0.0862	
SLR.F.Asc.Asc	Fucoids	0.0381	
SLR.F.Fspi	Fucoids	0.2687	
SLR.F.Fspi	Fucoids	0.0939	
SLR.F.Fspi	Fucoids	0.0025	Target Note 1
SLR.F.Fspi	Fucoids	0.0428	
SLR.F.Fves	Fucoids	0.1837	
SLR.F.Pel	Fucoids	0.0025	Target Note 1
SLR.FX.EphX	Algal turf	0.1499	

Stack rocks & St Govans

Biotope	Lifeform	Area (ha)	notes
EIR.KFaR.Ala.Ldig	Kelp	0.0087	
ELR.FR.Coff	Fucoids	0.0458	
ELR.MB.BPat	Mussels & Barnacles	0.4950	
ELR.MB.BPat.Sem	Mussels & Barnacles	0.7401	
ELR.MB.MytB	Mussels & Barnacles	0.0203	
LGS.S.AEur	Sand	0.6072	
LR.L.Ver.Ver	Lichens & algae	0.0388	
LR.L.Ver.Ver	Lichens & algae	0.0272	
LR.Rkp.Cor	Algal turf	0.0025	Target Note 78
LR.Rkp.G	Algal turf	0.0025	Target Note 9
MIR.KR.Ldig.Pid	Kelp	0.0423	
MLR.Eph.Ent	Algal turf	0.0025	Target Note 81
MLR.Eph.EntPor	Algal turf	0.0025	Target Note 57
MLR.Eph.EntPor	Algal turf	0.0024	
MLR.R.Pal	Algal turf	0.0025	Target Note 81
MLR.R.XR	Algal turf	0.1566	
MLR.R.XR	Algal turf	0.0316	
ELR.MB.BPat	Mussels & Barnacles	0.0015	
ELR.MB.BPat	Mussels & Barnacles	0.0024	
ELR.MB.BPat	Mussels & Barnacles	0.0024	
ELR.MB.BPat	Mussels & Barnacles	0.0037	
ELR.MB.BPat	Mussels & Barnacles	0.0024	
ELR.MB.BPat	Mussels & Barnacles	0.0044	
ELR.MB.BPat	Mussels & Barnacles	0.0524	
ELR.MB.BPat	Mussels & Barnacles	0.0057	
ELR.MB.BPat	Mussels & Barnacles	0.0025	Target Note 33
ELR.MB.BPat	Mussels & Barnacles	0.9566	
ELR.MB.BPat	Mussels & Barnacles	0.2322	
ELR.MB.BPat.Cat	Mussels & Barnacles	0.0080	
ELR.MB.BPat.Cat	Mussels & Barnacles	0.0025	Target Note 29
ELR.MB.BPat.Cat	Mussels & Barnacles	0.0025	Target Note 42
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0025	Target Note 16
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0151	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0104	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0025	Target Note 40
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0174	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0366	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0025	Target Note 10
ELR.MB.MytB	Mussels & Barnacles	0.0074	
ELR.MB.MytB	Mussels & Barnacles	0.0013	
ELR.MB.MytB	Mussels & Barnacles	0.0411	

Biotope	Lifeform	Area (ha)	notes
ELR.MB.MytB	Mussels & Barnacles	0.0025	Target Note 24
ELR.MB.MytB	Mussels & Barnacles	0.0253	
ELR.MB.MytB	Mussels & Barnacles	0.0106	
ELR.MB.MytB	Mussels & Barnacles	0.0025	Target Note 35
ELR.MB.MytB	Mussels & Barnacles	0.0025	Target Note 40
IR.FaSwV.AlcByH	Short faunal turf; crusts & cushions	0.0025	Target Note 1
LGS.S.AEur	Sand	0.0025	Target Note 38
LGS.S.AEur	Sand	0.3108	
LGS.S.AEur	Sand	0.2205	
LGS.S.AEur	Sand	0.1299	
LGS.S.AEur	Sand	0.2657	
LGS.S.AEur	Sand	0.1052	
LGS.S.AP.P	Sand	0.1938	
LGS.S.AP.P	Sand	0.0573	
LGS.S.Tal	Sand	0.0025	Target Note 18
LGS.Sh.BarSh	Shingle, coarse sand, mixed sediments	0.0189	
LGS.Sh.BarSh	Shingle, coarse sand, mixed sediments	0.0126	
LR.CvOv.BarCv	Lichens & algae	0.0025	Target Note 33
LR.CvOv.BarCv	Lichens & algae	0.0025	Target Note 5
LR.CvOv.BarCv	Lichens & algae	0.0025	Target Note 14
LR.CvOv.BarCv	Lichens & algae	0.0025	Target Note 19
LR.CvOv.BarCv	Lichens & algae	0.0025	Target Note 21
LR.CvOv.BarCv	Lichens & algae	0.0025	Target Note 29
LR.CvOv.FaCr	Short faunal turf; crusts & cushions	0.0025	Target Note 5
LR.CvOv.FaCr	Short faunal turf; crusts & cushions	0.0025	Target Note 14
LR.CvOv.FaCr	Short faunal turf; crusts & cushions	0.0025	Target Note 19
LR.CvOv.FaCr	Short faunal turf; crusts & cushions	0.0025	Target Note 21
LR.CvOv.SByAs.Cv	Short faunal turf; crusts & cushions	0.0025	Target Note 14
LR.CvOv.ScrFa	Short faunal turf; crusts & cushions	0.0025	Target Note 12
LR.CvOv.SpR.Cv	Short faunal turf; crusts & cushions	0.0025	Target Note 19
LR.CvOv.SpR.Cv	Short faunal turf; crusts & cushions	0.0025	Target Note 21
LR.CvOv.SpR.Cv	Short faunal turf; crusts & cushions	0.0025	Target Note 14
LR.CvOv.SpR.Ov	Short faunal turf; crusts & cushions	0.0025	Target Note 2
LR.CvOv.SpR.Ov	Short faunal turf; crusts & cushions	0.0025	Target Note 23
LR.CvOv.SpR.Ov	Short faunal turf; crusts & cushions	0.0025	Target Note 26
LR.CvOv.SpR.Ov	Short faunal turf; crusts & cushions	0.0025	Target Note 32
LR.CvOv.VmucHil	Lichens & algae	0.0025	Target Note 17
LR.CvOv.VmucHil	Lichens & algae	0.0025	Target Note 37
LR.L.Ver.Ver	Lichens & algae	0.0595	
LR.L.Ver.Ver	Lichens & algae	0.0127	
LR.L.Ver.Ver	Lichens & algae	0.0137	
LR.L.Ver.Ver	Lichens & algae	0.0433	
LR.L.Ver.Ver	Lichens & algae	0.0708	
LR.L.Ver.Ver	Lichens & algae	0.0327	
LR.L.Ver.Ver	Lichens & algae	0.0025	Target Note 40
LR.L.Ver.Ver	Lichens & algae	0.0012	
LR.L.YG	Lichens & algae	0.0041	
LR.L.YG	Lichens & algae	0.0051	
LR.L.YG	Lichens & algae	0.0210	
LR.L.YG	Lichens & algae	0.0699	
LR.L.YG	Lichens & algae	0.0344	
LR.L.YG	Lichens & algae	0.0025	Target Note 40
LR.L.YG	Lichens & algae	0.0320	

Biotope	Lifeform	Area (ha)	notes
LR.Rkp.FK	Fucoids	0.0025	Target Note 3
LR.Rkp.FK	Fucoids	0.0025	Target Note 15
LR.Rkp.FK	Fucoids	0.0025	Target Note 20
LR.Rkp.FK	Fucoids	0.0025	Target Note 31
LR.Rkp.SwSed	Fucoids	0.0025	Target Note 30
MIR.KR.Ldig.Ldig	Kelp	0.0432	
MIR.KR.Ldig.Pid	Kelp	0.0021	
MIR.KR.Ldig.Pid	Kelp	0.0033	
MIR.KR.Ldig.Pid	Kelp	0.0093	
MIR.KR.Ldig.Pid	Kelp	0.0231	
MIR.KR.Ldig.Pid	Kelp	0.0119	
MIR.KR.Ldig.Pid	Kelp	0.0059	
MIR.KR.Ldig.Pid	Kelp	0.0025	Target Note 6
MIR.KR.Ldig.Pid	Kelp	0.0004	
MIR.KR.Ldig.Pid	Kelp	0.0025	Target Note 24
MIR.KR.Ldig.Pid	Kelp	0.0043	
MIR.KR.Ldig.Pid	Kelp	0.0109	
MIR.KR.Ldig.Pid	Kelp	0.0025	Target Note 40
MLR.BF.Fser.Pid	Fucoids	0.0091	
MLR.BF.Fser.Pid	Fucoids	0.0448	
MLR.BF.Fser.Pid	Fucoids	0.0025	Target Note 40
MLR.BF.Fser.Pid	Fucoids	0.0025	Target Note 9
MLR.BF.Fser.R	Fucoids	0.0037	
MLR.BF.Fser.R	Fucoids	0.0025	Target note 32
MLR.BF.Fser.R	Fucoids	0.0013	
MLR.BF.Fser.R	Fucoids	0.0058	
MLR.BF.Fser.R	Fucoids	0.0125	
MLR.BF.Fser.R	Fucoids	0.0295	
MLR.BF.PelB	Fucoids	0.0025	Target Note 42
MLR.Eph.Rho	Algal turf	0.2153	
MLR.Eph.Rho	Algal turf	0.0025	Target Note 27
MLR.Eph.Rho	Algal turf	0.0363	
MLR.Eph.Rho	Algal turf	0.0040	
MLR.Eph.Rho	Algal turf	0.0039	
MLR.Eph.Rho	Algal turf	0.1060	
MLR.R.Pal	Algal turf	0.0025	Target Note 8
MLR.R.XR	Algal turf	0.0025	Target Note 30
MLR.R.XR	Algal turf	0.0025	Target Note 11

Broad Haven South

Biotope	Lifeform	Area (ha)	notes
EIR.SG.FoSwCC	Faunal and algal turf	0.0025	Target Note 69
ELR.MB.BPat	Mussels & Barnacles	0.0015	
ELR.MB.BPat	Mussels & Barnacles	0.0003	
ELR.MB.BPat	Mussels & Barnacles	0.0044	
ELR.MB.BPat	Mussels & Barnacles	0.0202	
ELR.MB.BPat	Mussels & Barnacles	0.9566	
ELR.MB.BPat	Mussels & Barnacles	0.1282	
ELR.MB.BPat	Mussels & Barnacles	0.0035	
ELR.MB.BPat	Mussels & Barnacles	0.1248	
ELR.MB.BPat	Mussels & Barnacles	0.0338	
ELR.MB.BPat	Mussels & Barnacles	0.0025	
ELR.MB.BPat	Mussels & Barnacles	0.0020	
ELR.MB.BPat	Mussels & Barnacles	0.0001	

Biotope	Lifeform	Area (ha)	notes
ELR.MB.BPat.Cat	Mussels & Barnacles	0.0025	Target Note 63
ELR.MB.BPat.Cat	Mussels & Barnacles	0.0134	
ELR.MB.BPat.Cat	Mussels & Barnacles	0.0025	Target Note 42
ELR.MB.BPat.Cat	Mussels & Barnacles	0.0025	Target Note 46
ELR.MB.BPat.Cht	Mussels & Barnacles	0.0282	
ELR.MB.BPat.Cht	Mussels & Barnacles	0.0338	
ELR.MB.BPat.Cht	Mussels & Barnacles	0.0105	
ELR.MB.BPat.Cht	Mussels & Barnacles	0.0033	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0025	Target Note 40
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0098	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0012	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0307	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0517	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0055	
ELR.MB.BPat.Sem	Mussels & Barnacles	0.2645	
ELR.MB.BPat.Sem	Mussels & Barnacles	0.0538	
ELR.MB.BPat.Sem	Mussels & Barnacles	0.1152	
ELR.MB.MytB	Mussels & Barnacles	0.0027	
ELR.MB.MytB	Mussels & Barnacles	0.0007	
ELR.MB.MytB	Mussels & Barnacles	0.0074	
ELR.MB.MytB	Mussels & Barnacles	0.0189	
ELR.MB.MytB	Mussels & Barnacles	0.0309	
ELR.MB.MytB	Mussels & Barnacles	0.0013	
ELR.MB.MytB	Mussels & Barnacles	0.0025	Target Note 40
ELR.MB.MytB	Mussels & Barnacles	0.0464	
ELR.MB.MytB	Mussels & Barnacles	0.0103	
ELR.MB.MytB	Mussels & Barnacles	0.0245	
ELR.MB.MytB	Mussels & Barnacles	0.0647	
ELR.MB.MytB	Mussels & Barnacles	0.0088	
ELR.MB.MytB	Mussels & Barnacles	0.0088	
ELR.MB.MytB	Mussels & Barnacles	0.0006	
LGS.S.AEur	Sand	0.3108	
LGS.S.AEur	Sand	0.2339	
LGS.S.AEur	Sand	0.0956	
LGS.S.AEur	Sand	8.6036	
LGS.S.AEur	Sand	0.0839	
LGS.S.AEur	Sand	0.2205	
LGS.S.BarSnd	Sand	1.7611	
LGS.S.Tal	Sand	0.1058	
LGS.S.Tal	Sand	1.0599	
LGS.Sh.BarSh	Shingle, coarse sand, mixed sediments	0.0025	Target Note 54
LGS.Sh.BarSh	Shingle, coarse sand, mixed sediments	0.0770	
LGS.Sh.BarSh	Shingle, coarse sand, mixed sediments	0.0760	
LR.CvOv.AudCla	Lichens & algae	0.0025	Target note 62
LR.CvOv.AudCla	Lichens & algae	0.0025	Target Note 62
LR.CvOv.BarCv	Lichens & algae	0.0025	Target Note 50
LR.CvOv.BarCv	Lichens & algae	0.0025	Target Note 68
LR.CvOv.BarCv	Lichens & algae	0.0025	Target Note 72
LR.CvOv.FaCr	Short faunal turf; crusts & cushions	0.0025	Target Note 73
LR.CvOv.FaCr	Short faunal turf; crusts & cushions	0.0025	Target Note 43
LR.CvOv.SByAs.Ov	Short faunal turf; crusts & cushions	0.0025	Target Note 68
LR.CvOv.ScrFa	Short faunal turf; crusts & cushions	0.0025	Target Note 43

Biotope	Lifeform	Area (ha)	notes
LR.CvOv.ScrFa	Short faunal turf; crusts & cushions	0.0025	Target Note 47
LR.CvOv.ScrFa	Short faunal turf; crusts & cushions	0.0025	Target Note 65
LR.CvOv.ScrFa	Short faunal turf; crusts & cushions	0.0025	Target Note 72
LR.CvOv.SpR.Ov	Short faunal turf; crusts & cushions	0.0025	Target Note 52
LR.CvOv.SpR.Ov	Short faunal turf; crusts & cushions	0.0025	Target Note 53
LR.L.Ver.Ver	Lichens & algae	0.0108	
LR.L.Ver.Ver	Lichens & algae	0.0127	
LR.L.Ver.Ver	Lichens & algae	0.0137	
LR.L.Ver.Ver	Lichens & algae	0.0025	Target Note 40
LR.L.Ver.Ver	Lichens & algae	0.1835	
LR.L.Ver.Ver	Lichens & algae	0.3193	
LR.L.Ver.Ver	Lichens & algae	0.0263	
LR.L.Ver.Ver	Lichens & algae	0.0126	
LR.L.Ver.Ver	Lichens & algae	0.0012	
LR.L.Ver.Ver	Lichens & algae	0.1869	
LR.L.YG	Lichens & algae	0.0027	
LR.L.YG	Lichens & algae	0.0041	
LR.L.YG	Lichens & algae	0.0051	
LR.L.YG	Lichens & algae	0.0025	Target Note 40
LR.L.YG	Lichens & algae	0.1802	
LR.L.YG	Lichens & algae	0.5446	
LR.L.YG	Lichens & algae	0.0164	
LR.L.YG	Lichens & algae	0.0119	
LR.L.YG	Lichens & algae	0.1238	
LR.Rkp.Cor	Algal turf	0.0025	Target Note 66
LR.Rkp.FK	Fucoids	0.0025	Target Note 70
LR.Rkp.G	Algal turf	0.0025	Target Note 64
MIR.KR.Ldig.Ldig	Kelp	0.0025	Target Note 69
MIR.KR.Ldig.Pid	Kelp	0.0007	
MIR.KR.Ldig.Pid	Kelp	0.0022	
MIR.KR.Ldig.Pid	Kelp	0.0007	
MIR.KR.Ldig.Pid	Kelp	0.0161	
MIR.KR.Ldig.Pid	Kelp	0.0096	
MIR.KR.Ldig.Pid	Kelp	0.0231	
MIR.KR.Ldig.Pid	Kelp	0.0025	Target Note 40
MIR.KR.Ldig.Pid	Kelp	0.0361	
MIR.KR.Ldig.Pid	Kelp	0.0516	
MIR.KR.Ldig.Pid	Kelp	0.0119	
MLR.BF.Fser.Fser	Fucoids	0.0025	Target Note 67
MLR.BF.Fser.Pid	Fucoids	0.0397	
MLR.BF.Fser.Pid	Fucoids	0.0079	
MLR.BF.Fser.Pid	Fucoids	0.0025	Target Note 40
MLR.BF.Fser.Pid	Fucoids	0.0587	
MLR.BF.Fser.Pid	Fucoids	0.0254	
MLR.BF.Fser.Pid	Fucoids	0.0104	
MLR.BF.Fser.Pid	Fucoids	0.0005	
MLR.BF.Fser.Pid	Fucoids	0.0020	
MLR.BF.Fser.Pid	Fucoids	0.0007	
MLR.BF.Fser.R	Fucoids	0.0031	
MLR.BF.Fser.R	Fucoids	0.0075	
MLR.BF.Fser.R	Fucoids	0.0125	
MLR.BF.FvesB	Fucoids	0.0025	Target Note 56
MLR.BF.FvesB	Fucoids	0.0108	
MLR.BF.PelB	Fucoids	0.0025	Target Note 56
MLR.BF.PelB	Fucoids	0.0025	Target Note 42

Biotope	Lifeform	Area (ha)	notes
MLR.Eph.Ent	Algal turf	0.0025	Target Note 54
MLR.Eph.Ent	Algal turf	0.0025	Target Note 67
MLR.Eph.EntPor	Algal turf	0.0345	
MLR.Eph.EntPor	Algal turf	0.0665	
MLR.Eph.EntPor	Algal turf	0.0108	
MLR.Eph.Rho	Algal turf	0.2153	
MLR.Eph.Rho	Algal turf	0.0033	
MLR.Eph.Rho	Algal turf	0.0040	
MLR.Eph.Rho	Algal turf	0.0035	
MLR.Eph.Rho	Algal turf	0.0099	
MLR.Eph.Rho	Algal turf	0.1060	
MLR.MF.MytFR	Mussels & Barnacles	0.0817	
MLR.MF.MytFR	Mussels & Barnacles	0.0013	
MLR.R.Osm	Algal turf	0.0025	Target Note 45
MLR.R.XR	Algal turf	0.0025	Target Note 69
SLR.F.Asc.Asc	Fucoids	0.0092	
SLR.F.Fspi	Fucoids	0.0234	
SLR.F.Fspi	Fucoids	0.0355	
SLR.F.Pel	Fucoids	0.0019	
SLR.F.Pel	Fucoids	0.0025	Target Note 61
SLR.F.Pel	Fucoids	0.0050	

Barafundle Bay

Biotope	Lifeform	Area (ha)	notes
ELR.MB.BPat	Mussels & Barnacles	0.0053	
ELR.MB.BPat	Mussels & Barnacles	0.2577	
ELR.MB.BPat.Cat	Mussels & Barnacles	0.0025	Target Note 14
ELR.MB.BPat.Cht	Mussels & Barnacles	0.0338	
ELR.MB.BPat.Cht	Mussels & Barnacles	0.0012	
ELR.MB.BPat.Cht	Mussels & Barnacles	0.0167	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0413	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0837	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0025	Target Note 14
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0025	Target Note 18
ELR.MB.BPat.Sem	Mussels & Barnacles	0.0426	
ELR.MB.BPat.Sem	Mussels & Barnacles	0.2448	
ELR.MB.BPat.Sem	Mussels & Barnacles	0.0080	
ELR.MB.BPat.Sem	Mussels & Barnacles	0.0027	
ELR.MB.MytB	Mussels & Barnacles	0.0166	
LGS.S.AEur	Sand	2.1706	
LGS.S.AEur	Sand	0.0025	Target Note 24
LGS.S.AEur	Sand	0.2062	
LGS.S.AP.P	Sand	1.2651	
LGS.S.BarSnd	Sand	0.9356	
LGS.S.Tal	Sand	0.4462	
LR.CvOv.BarCv	Lichens & algae	0.0025	Target Note 22
LR.CvOv.BarCv	Lichens & algae	0.0025	Target Note 24
LR.CvOv.BarCv	Lichens & algae	0.0025	Target Note 25
LR.CvOv.BarCv	Lichens & algae	0.0025	Target Note 25
LR.CvOv.SByAs.Cv	Short faunal turf; crusts & cushions	0.0025	Target Note 11
LR.CvOv.SByAs.Ov	Short faunal turf; crusts & cushions	0.0025	Target Note 25
LR.CvOv.SByAs.Ov	Short faunal turf; crusts & cushions	0.0025	Target Note 25
LR.CvOv.ScrFa	Short faunal turf; crusts & cushions	0.0025	Target Note 11

Biotope	Lifeform	Area (ha)	notes
LR.CvOv.ScrFa	Short faunal turf; crusts & cushions	0.0025	Target Note 24
LR.CvOv.SpR.Cv	Short faunal turf; crusts & cushions	0.0025	Target Note 11
LR.CvOv.SpR.Ov	Short faunal turf; crusts & cushions	0.0025	Target Note 32
LR.CvOv.SpR.Ov	Short faunal turf; crusts & cushions	0.0025	Target Note 11
LR.CvOv.SpR.Ov	Short faunal turf; crusts & cushions	0.0025	Target Note 13
LR.CvOv.SpR.Ov	Short faunal turf; crusts & cushions	0.0025	Target Note 28
LR.CvOv.SpR.Ov	Short faunal turf; crusts & cushions	0.0025	Target Note 28
LR.CvOv.VmucHil	Lichens & algae	0.0025	Target Note 11
LR.CvOv.VmucHil	Lichens & algae	0.0025	Target Note 22
LR.CvOv.VmucHil	Lichens & algae	0.0025	Target Note 24
LR.L.Ver.Ver	Lichens & algae	0.2242	
LR.L.YG	Lichens & algae	0.2892	
LR.L.YG	Lichens & algae	0.2253	
LR.L	Lichens & algae	0.0114	
LR.Rkp.Cor	Algal turf	0.0025	Target Note 20
MIR.KR.Ldig.Ldig	Kelp	0.0027	
MIR.KR.Ldig.Pid	Kelp	0.0056	
MIR.KR.Ldig.Pid	Kelp	0.0271	
MLR.BF.Fser.Pid	Fucoids	0.0046	
MLR.BF.Fser.Pid	Fucoids	0.0543	
MLR.BF.Fser.Pid	Fucoids	0.0200	
MLR.BF.Fser.R	Fucoids	0.0223	
MLR.BF.Fser.R	Fucoids	0.0034	
MLR.Eph.EntPor	Algal turf	0.1385	
MLR.Eph.EntPor	Algal turf	0.0207	
MLR.Eph.EntPor	Algal turf	0.1932	
MLR.Eph.Rho	Algal turf	0.0025	Target Note 30
MLR.Eph.Rho	Algal turf	0.0013	
MLR.R.Osm	Algal turf	0.0168	
MLR.R.XR	Algal turf	0.0025	Target Note 30
MLR.R.XR	Algal turf	0.0074	
SLR.F.Fves	Fucoids	0.0122	
SLR.F.Pel	Fucoids	0.0025	Target Note 17

Stackpole Quay

Biotope	Lifeform	Area (ha)	notes
ELR.MB.BPat	Mussels & Barnacles	0.2577	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0837	
LR.CvOv.ScrFa	Short faunal turf; crusts & cushions	0.0025	Target Note 37
LR.CvOv.SpR.Cv	Short faunal turf; crusts & cushions	0.0025	Target Note 34
LR.CvOv.SpR.Ov	Short faunal turf; crusts & cushions	0.0025	Target Note 32
LR.CvOv.SpR.Ov	Short faunal turf; crusts & cushions	0.0025	Target Note 34
LR.CvOv.SpR.Ov	Short faunal turf; crusts & cushions	0.0025	Target Note 36
LR.L.Ver.Ver	Lichens & algae	0.2910	
LR.L.Ver.Ver	Lichens & algae	0.0032	
LR.L.YG	Lichens & algae	0.2892	
MIR.KR.Ldig.Pid	Kelp	0.0671	
MLR.BF.Fser.Pid	Fucoids	0.0457	
MLR.BF.Fser.Pid	Fucoids	0.0200	
MLR.BF.Fser.R	Fucoids	0.0223	
MLR.Eph.EntPor	Algal turf	0.0207	
MLR.Eph.Rho	Algal turf	0.0025	Target Note 30
MLR.R.XR	Algal turf	0.0025	Target Note 30
MLR.R.XR	Algal turf	0.0074	

Biotope	Lifeform	Area (ha)	notes
ELR.MB.BPat	Mussels & Barnacles	1.0224	
ELR.MB.BPat	Mussels & Barnacles	0.0847	
ELR.MB.BPat.Cat	Mussels & Barnacles	0.0030	
ELR.MB.BPat.Cht	Mussels & Barnacles	0.0434	
ELR.MB.BPat.Cht	Mussels & Barnacles	0.0025	Target Note 5
ELR.MB.BPat.Cht	Mussels & Barnacles	0.0638	
ELR.MB.BPat.Cht	Mussels & Barnacles	0.0112	
ELR.MB.BPat.Cht	Mussels & Barnacles	0.0224	
ELR.MB.BPat.Cht	Mussels & Barnacles	1.3088	
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0025	Target Note 11
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0025	Target Note 23
ELR.MB.BPat.Lic	Mussels & Barnacles	0.0014	
ELR.MB.BPat.Sem	Mussels & Barnacles	0.0083	
ELR.MB.BPat.Sem	Mussels & Barnacles	0.0371	
ELR.MB.BPat.Sem	Mussels & Barnacles	0.1030	
ELR.MB.BPat.Sem	Mussels & Barnacles	0.0025	Target Note 5
ELR.MB.BPat.Sem	Mussels & Barnacles	0.0273	
ELR.MB.BPat.Sem	Mussels & Barnacles	3.9502	
ELR.MB.BPat.Sem	Mussels & Barnacles	0.0038	
ELR.MB.BPat.Sem	Mussels & Barnacles	0.0034	
ELR.MB.BPat.Sem	Mussels & Barnacles	0.0428	
ELR.MB.BPat.Sem	Mussels & Barnacles	0.0026	
ELR.MB.BPat.Sem	Mussels & Barnacles	0.0428	
LGS.S.AP.P	Sand	0.0962	
LGS.S.AP.P	Sand	0.0741	
LGS.S.AP.P	Sand	0.0909	
LGS.S.AP.P	Sand	0.1475	
LGS.S.Tal	Sand	0.0223	
LGS.S.Tal	Sand	0.0085	
LGS.Sh.BarSh	Shingle, coarse sand, mixed sediments	0.1201	
LGS.Sh.BarSh	Shingle, coarse sand, mixed sediments	0.0094	
LGS.Sh.BarSh	Shingle, coarse sand, mixed sediments	0.0152	
LGS.Sh.BarSh	Shingle, coarse sand, mixed sediments	0.1641	
LGS.Sh.BarSh	Shingle, coarse sand, mixed sediments	0.0542	
LGS.Sh.BarSh	Shingle, coarse sand, mixed sediments	0.0852	
LGS.Sh.BarSh	Shingle, coarse sand, mixed sediments	0.0426	
LGS.Sh.BarSh	Shingle, coarse sand, mixed sediments	0.0657	
LR.CvOv.SByAs.Ov	Short faunal turf; crusts & cushions	0.0025	Target Note 17
LR.CvOv.SR.Den.Cv	Short faunal turf; crusts & cushions	0.0025	Target Note 21
LR.CvOv.SpR.Ov	Short faunal turf; crusts & cushions	0.0025	Target Note 18
LR.CvOv.SpR.Ov	Short faunal turf; crusts & cushions	0.0025	Target Note 27
LR.CvOv.SpR.Ov	Short faunal turf; crusts & cushions	0.0025	Target Note 29
LR.CvOv.SpR.Ov	Short faunal turf; crusts & cushions	0.0025	Target Note 29
LR.CvOv.SpR.Ov	Short faunal turf; crusts & cushions	0.0025	Target Note 32
LR.CvOv.VmucHil	Lichens & algae	0.0025	Target Note 16
LR.L.Ver.Ver	Lichens & algae	0.0102	
LR.L.Ver.Ver	Lichens & algae	0.0164	

Biotope	Lifeform	Area (ha)	notes
LR.L.Ver.Ver	Lichens & algae	0.0426	
LR.L.Ver.Ver	Lichens & algae	0.0315	
LR.L.Ver.Ver	Lichens & algae	0.0371	
LR.L.Ver.Ver	Lichens & algae	0.0025	Target Note 5
LR.L.Ver.Ver	Lichens & algae	0.0844	
LR.L.Ver.Ver	Lichens & algae	0.0203	
LR.L.Ver.Ver	Lichens & algae	0.0089	
LR.L.Ver.Ver	Lichens & algae	0.0184	
LR.L.Ver.Ver	Lichens & algae	0.0133	
LR.L.Ver.Ver	Lichens & algae	0.0035	
LR.L.Ver.Ver	Lichens & algae	0.0029	
LR.L.YG	Lichens & algae	0.0034	
LR.L.YG	Lichens & algae	0.0096	
LR.L.YG	Lichens & algae	0.0140	
LR.L.YG	Lichens & algae	0.0305	
LR.L.YG	Lichens & algae	0.0232	
LR.L.YG	Lichens & algae	0.0268	
LR.L.YG	Lichens & algae	0.0025	Target Note 5
LR.L.YG	Lichens & algae	0.0254	
LR.L.YG	Lichens & algae	0.0047	
LR.L.YG	Lichens & algae	0.0655	
LR.L.YG	Lichens & algae	0.0088	
LR.L.YG	Lichens & algae	0.0036	
LR.Rkp.Cor	Algal turf	0.0025	Target Note 24
LR.Rkp.FK	Fucoids	0.0025	Target Note 1
LR.Rkp.FK	Fucoids	0.0025	Target Note 28
LR.Rkp.G	Algal turf	0.0025	Target Note 13
LR.Rkp.SwSed	Fucoids	0.0025	Target Note 22
MIR.KR.Ldig.Ldig	Kelp	0.4658	
MIR.KR.Ldig.Ldig	Kelp	0.0624	
MIR.KR.Ldig.Pid	Kelp	0.0025	Target Note 14
MLR.BF.Fser.Fser	Fucoids	0.1893	
MLR.BF.Fser.Fser	Fucoids	0.2684	
MLR.BF.Fser.R	Fucoids	0.1001	
MLR.BF.Fser.R	Fucoids	0.0077	
MLR.Eph.EntPor	Algal turf	0.0025	Target Note 31
MLR.Eph.Rho	Algal turf	0.0025	Target Note 25
MLR.R.Mas	Algal turf	0.0118	
MLR.R.Mas	Algal turf	0.1983	
MLR.R.Osm	Algal turf	0.0025	Target Note 5
MLR.R.Osm	Algal turf	0.0025	Target Note 26
SLR.F.Asc.Asc	Fucoids	0.0012	
SLR.F.Fspi	Fucoids	0.0025	Target Note 32
SLR.F.Fspi	Fucoids	0.0031	
SLR.F.Fspi	Fucoids	0.0529	
SLR.F.Pel	Fucoids	0.0014	
SLR.F.Pel	Fucoids	0.0324	
SLR.F.Pel	Fucoids	0.0031	
SLR.FX.BLlit	Fucoids	0.0423	
SLR.FX.EphX	Algal turf	0.0722	

Freshwater East

Biotope	Lifeform	Area (ha)	notes
ELR.MB.BPat	Mussels & Barnacles	0.0012	
ELR.MB.MytB	Mussels & Barnacles	0.0024	
ELR.MB.MytB	Mussels & Barnacles	0.0119	
ELR.MB.MytB	Mussels & Barnacles	0.0016	
ELR.MB.MytB	Mussels & Barnacles	0.0112	
ELR.MB.MytB	Mussels & Barnacles	0.0030	
ELR.MB.MytB	Mussels & Barnacles	0.1355	
LGS.S.AEur	Sand	9.8715	
LGS.S.AP.P	Sand	4.7534	
LGS.S.AP.P	Sand	0.2014	
LGS.S.BarSnd	Sand	0.6593	
LGS.S.BarSnd	Sand	2.0067	
LGS.S.Tal	Sand	0.2320	
LGS.S.Tal	Sand	0.9438	
LGS.Sh.BarSh	Shingle, coarse sand, mixed sediments	1.1940	
LGS.Sh.BarSh	Shingle, coarse sand, mixed sediments	0.3438	
LMU.Sm	Saltmarsh	0.0025	Target Note 15
LR.L.YG	Lichens & algae	0.0274	
MIR.SedK.LsacChoR	Kelp	0.0817	
SLR.FX.EphX	Algal turf	0.0921	
SLR.FX.EphX	Algal turf	0.2530	
SLR.FX.EphX	Algal turf	0.9013	
SLR.FX.EphX	Algal turf	0.4919	
SLR.FX.EphX	Algal turf	0.1263	
SLR.FX.EphX	Algal turf	0.1190	
SLR.FX.EphX	Algal turf	0.1487	

Appendix 2. Sensitivity assessment rationale - a summary

Assessing the sensitivity of species

The assessment process involves judging the intolerance of a species to change in an external factor arising from human activities or natural events. The rationale then assesses the likely recoverability of the species following cessation on the human activity or natural event. Intolerance and recoverability are then combined to provide a meaningful assessment of their overall sensitivity to environmental change.

1. Collate the key information for the species. The best available scientific information required to describe the biology and likely sensitivity of the species is collated using the resources of the National Marine Biological Library (NMBL), the World Wide Web, and the expertise of marine biologists based at the Marine Biological Association of the UK (MBA), Plymouth.

2. Indicate quality of available data. The *MarLIN* programme operates an internal quality

> **Box A2.1. Core definitions**
>
> **'Biotope'** refers to the combination of physical environment (habitat) and its distinctive assemblage of conspicuous species. For practical reasons of interpretation of terms used in directives, statutes and conventions, in some documents, 'biotope' is sometimes synonymized with 'habitat'.
>
> **'Habitat'** the place in which a plant or animal lives. It is defined for the marine environment according to geographical location, physiographic features and the physical and chemical environment (including salinity, wave exposure, strength of tidal streams, geology, biological zone, substratum), 'features' (such as crevices, overhangs, or rockpools) and 'modifiers' (for example sand-scour, wave-surge, or substratum mobility).
>
> **'Community'** refers to a group of organisms occurring in a particular environment, presumably interacting with each other and with the environment, and identifiable by means of ecological survey from other groups. The community is usually considered the biotic element of a biotope.
>
> **'Intolerance'** is the susceptibility of a habitat, community, or species (i.e. the components of a biotope) to damage, or death, from an external factor. Intolerance must be assessed relative to specified change in a specific environmental factor.
>
> **'Recoverability'** is the ability of a habitat, community, or species (i.e. the components of a biotope) to return to a state close to that which existed before the activity or event caused change.
>
> **'Sensitivity'** is dependent on the intolerance of a species or habitat to damage from an external factor and the time taken for its subsequent recovery. For example, a "highly sensitive" species or habitat is one that is very adversely affected by an external factor arising from human activities or natural events (killed/destroyed, 'high' intolerance) and is expected to recover only over a very long period of time, (10 to 25 years: 'low' recoverability). Intolerance and hence sensitivity must be assessed relative to a specified change in a specific environmental factor.

assurance procedure, to ensure that only the most accurate available information is provided on-line. The quality of the available evidence and our confidence in our assessments (based on availability of information) is clearly stated (see Table A2.1).

3. Assess the intolerance of the species to change in environmental factors. The likely intolerance (Table A2.2) of the species is assessed with respect to a specified magnitude and duration of change (the standard benchmark) for 24 separate environmental factors (see Table A2.3).

Precedence is given to direct evidence of effect or impact. For example, information from targeted studies / experiments that looked at the effect of the specific factor on the species, or targeted work / experiments on the effects of similar factors on similar species or studies of the likely effects of a factor. The assessment of intolerance (Table A2.2) is then made by reference to the reported change in environmental factors and their impact, relative to the magnitude and duration of the standard benchmarks and other relevant key information.

Table A2.1. Scale used to rank the level of information available to support the assessment of intolerance and recoverability

EVIDENCE / CONFIDENCE The scale indicates an appraisal of the specificity of the information (data) available to support the assessment of intolerance and recoverability.	
Rank	**Definition (adapted from Hiscock *et al.,* 1999)**
High	Assessment has been derived from sources that specifically deal with sensitivity and recoverability to a particular factor. Experimental work has been done investigating the effects of such a factor.
Moderate	Assessment has been derived from sources that consider the likely effects of a particular factor.
Low	Assessment has been derived from sources that only cover aspects of the biology of the species or from a general understanding of the species. No information is present regarding the effects of factors.
Very low	Assessment derived by 'informed judgement' where very little information is present at all on the species.
Not relevant	The available information does not support an assessment, the data is deficient, or no relevant information has been found.
Note: In some cases it is possible for limited evidence to be considered 'high' for the assessment of sensitivity to a specific factor. For example, if a species is known to lack eyes (or equivalent photoreceptors) then it could confidently be considered 'not sensitive' to visual disturbance and the level of evidence would be recorded as 'high'.	

In the absence of direct evidence, the *MarLIN* rationale includes simple decision trees to aid intolerance and recoverability assessment based on the available key information for the species. The decision trees provide a systematic and transparent approach to assessment. The decision trees are described in full by Tyler-Walters *et al.* (2001).

Table A2.2. Species intolerance (previously 'sensitivity' and revised April 2003).

SPECIES INTOLERANCE The susceptibility of a species population to damage, or death, from an external factor. Intolerance is assessed relative to change in a specific factor.	
Rank	**Definition**
High	The species population is likely to be killed/destroyed by the factor under consideration.
Intermediate	Some individuals of the species may be killed/destroyed by the factor under consideration and the viability of a species population may be reduced.
Low	The species population will not be killed/destroyed by the factor under consideration. However, the viability of a species population may be reduced.
Tolerant	The factor does not have a detectable effect on survival or viability of a species.
Tolerant*	Population of a species may increase in abundance or biomass as a result of the factor.
Not relevant	This rating applies to species where the factor is not relevant because they are protected from the factor (for instance, through a burrowing habit), or can move away from the factor.

Table A2.3. Environmental factors for which intolerance and hence sensitivity is assessed.

Physical factors	
	Substratum loss
	Smothering
	Suspended sediment
	Desiccation
	Changes in emergence regime
	Changes in water flow rate
	Changes in temperature
	Changes in turbidity
	Changes in wave exposure
	Noise
	Visual presence
	Abrasion and physical disturbance
	Displacement
Chemical factors	
	Synthetic compounds
	Heavy metals
	Hydrocarbons
	Radionuclides
	Changes in nutrient levels
	Changes in salinity
	Changes in oxygenation
Biological factors	
	Introduction of microbial pathogens
	Introduction of non-native species and
	Selective extraction of this species
	Selective extraction of other species

4. Assess the recoverability of the species. The likely recoverability of a species from disturbance or damage is dependent on its ability to regenerate, regrow, recruit or recolonize, depending on the extent of damage incurred and hence its intolerance. The recoverability of a species is assessed against the recoverability scale (Table A3.4) by reference to direct evidence of recruitment, recolonization or recovery (e.g. after environmental impact or experimental manipulation in the field) and/or key information on the reproductive biology, habitat preferences and distribution of the species.

5. Assess the sensitivity of the species. The overall sensitivity rank is derived from the combination of intolerance and recoverability using the rationale shown in Tables A2.5 and A2.6 below.

The sensitivity assessment rationale uses the question 'does it matter if……?', together with the definitions of sensitive habitats and species proposed in the Review of Marine Nature Conservation (Laffoley *et al.*, 2000) as touch-stones throughout. Due to the importance of recoverability in assessing the continued survival of a habitat or species population, the scale is intuitively weighted towards recoverability. However, where recovery is likely to occur in a short period of time, intolerance has been given a greater weight rather than under-estimate the potential sensitivity of marine habitats and species. The sensitivity scales and definitions are designed to be meaningful in marine environmental management, protection, and conservation.

Table A2.4. Recoverability.

RECOVERABILITY The ability of a habitat, community, or individual (or individual colony) of species to redress damage sustained as a result of an external factor.	
Recoverability is only applicable if and when the impacting factor has been removed or has stopped. Ranks also only refer to the recoverability potential of a species, based on their reproductive biology etc.	
Rank	**Definition (From Hiscock *et al*. 1999)**
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 5 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 6 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’.

For example, 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 1 year or up to 5 years ('very high' or 'high' recoverability) then it would be considered to be of low sensitivity. The scenarios used to derive the sensitivity scale are listed in Table A2.6.

Table A2.5. Defining 'sensitivity' *sensu lato* for habitats and species. **='Reduced viability' includes physiological stress, reduced fecundity, reduced growth, and partial death of a colonial animal or plant.

Sensitivity scale	Sensitivity definition or scenario
Very High	'Very high' sensitivity is indicated by the following scenario: • The habitat or species is very adversely affected by an external factor arising from human activities or natural events (either killed/destroyed, 'high' intolerance) and is expected to recover only over a prolonged period of time, i.e. >25 years or not at all (recoverability is 'very low' or 'none'). • The habitat or species is adversely affected by an external factor arising from human activities or natural events (damaged, 'intermediate' intolerance) but is not expected to recover at all (recoverability is 'none').
High	'High' sensitivity is indicated by the following scenarios: • The 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). • The habitat or species is adversely affected by an external factor arising from human activities or natural events (damaged, 'intermediate' intolerance) and is expected to recover over a very long period of time, i.e. >10 years (recoverability is 'low', or 'very low'). • The habitat or species is affected by an external factor arising from human activities or natural events (reduced viability **, 'low' intolerance) but is not expected to recover at all (recoverability is 'none'), so that the habitat or species may be vulnerable to subsequent damage.
Moderate	'Moderate' sensitivity is indicated by the following scenarios: • The habitat or species is very adversely affected by an external factor arising from human activities or natural events (killed/destroyed, 'high' intolerance) but is expected to take more than 1 year or up to 10 years to recover ('moderate' or 'high' recoverability). • The habitat or species is adversely affected by an external factor arising from human activities or natural events (damaged, 'intermediate' intolerance) and is expected to recover over a long period of time, i.e. >5 or up to 10 years ('moderate' recoverability). • The habitat or species is affected by an external factor arising from human activities or natural events (reduced viability **, 'low' intolerance) but is expected to recover over a very long period of time, i.e. >10 years (recoverability is 'low', 'very low'), during which time the habitat or species may be vulnerable to subsequent damage.

Table A2.5. Defining 'sensitivity' *sensu lato* for habitats and species (continued). **='Reduced viability' includes physiological stress, reduced fecundity, reduced growth, and partial death of a colonial animal or plant.

Low	'Low' sensitivity is indicated by the following scenarios: • The habitat or species is very adversely affected by an external factor arising from human activities or natural events (killed/destroyed, 'high' intolerance) but is expected to recover rapidly, i.e. within 1 year ('very high' recoverability). • The 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 1 year or up to 5 years ('very high' or 'high' recoverability). • The habitat or species is affected by an external factor arising from human activities or natural events (reduced viability **, 'low' intolerance) but is expected to take more than 1 year or up to 10 years to recover ('moderate' or 'high' recoverability).
Very low	'Very low' is indicated by the following scenarios: • The habitat or species is very adversely affected by an external factor arising from human activities or natural events (killed/destroyed, 'high' intolerance) but is expected to recover rapidly i.e. within a week ('immediate' recoverability). • The 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 rapidly, i.e. within a week ('immediate' recoverability). • The habitat or species is affected by an external factor arising from human activities or natural events (reduced viability **, 'low' intolerance) but is expected to recover within a year ('very high' recoverability).
Not sensitive	'Not sensitive' is indicated by the following scenarios: • The habitat or species is affected by an external factor arising from human activities or natural events (reduced viability **, 'low' intolerance) but is expected to recover rapidly, i.e. within a week ('immediate' recoverability). • The habitat or species is tolerant of changes in the external factor.
Not sensitive*	The habitat or species may benefit from the change in an external factor (intolerance has been assessed as 'tolerant*').
Not relevant	The habitat or species is protected from changes in an external factor (i.e. through a burrowing habit or depth), or is able to avoid the external factor.

NB: Where there is insufficient information to assess the recoverability of a habitat or species ('insufficient information') the 'precautionary principle' will be used and the 'recovery' *will be assumed* to take a very long time i.e. 'low' recoverability in the derivation of a sensitivity rank.

The above definitions and scenarios give rise to the decision matrix shown in Table 6. The decision matrix is used to automate the combination of 'intolerance' and 'recoverability' within the *MarLIN* biology and sensitivity database.

The decision matrix shown in Table A3.6 is not symmetrical because the scale represents scenarios in which the potential damage to the species or habitat 'matters'. The scale is intuitively weighted towards recoverability, although in a few cases intolerance has been given a greater weight rather than under-estimate the potential sensitivity of marine habitats and species.

Table A2.6. Combining 'intolerance' and 'recoverability' assessments to determine 'sensitivity'. NS = not sensitive, NR = not relevant.

		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)
Intolerance	*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*

Please note that the intolerance, recoverability and sensitivity ranks should be read in conjunction with the on-line rationale for each assessment, which outline the evidence and key information used and any judgements made in the assessment. The information used and evidence collated is fully referenced throughout.

6. Signing-off. *MarLIN* reviews are checked by the Programme Director for accuracy and clarity and the required changes made before the review goes ‘on-line’ on the Web site.

7. Referee. As a final stage in the *MarLIN* quality assurance, Key Information reviews are subject to peer review by an external marine biologist where possible.

Assessing the sensitivity of habitats and their associated species (biotopes)

The *MarLIN* approach to the assessment of the sensitivity of biotopes assumes that the sensitivity of a community within a biotope is dependent upon and, therefore, is indicated by the sensitivity of the species within that community. Species that indicate the sensitivity of a biotope are identified as those species that significantly influence the ecology of that component community (see Table A2.7). The loss of one or more of these species would result in changes in the population(s) of associated species and their interactions. The criteria used to identify species that indicate biotope sensitivity subdivide species into ‘key’ and ‘important’ based on the likely magnitude of the resultant change.

The protocol used to prepare a review of the biology and sensitivity key information for a biotope is given below.

1. Collate key information on the biotope. The best available scientific information required to describe the ecology and likely sensitivity of the biotope is collated using the resources of the National Marine Biological Library (NMBL), the World Wide Web, and the expertise of marine biologists based at the MBA, Plymouth.

2. Select species indicative of biotope sensitivity. Species are selected based on the review of the ecology of habitat and community, where direct evidence of community interaction or dependency is available, or where the species are ‘important characterizing’ (Table A2.7).

3. Review key information for the selected species. Key information on the biology and sensitivity of the indicative species is researched.

4. Indicate quality of available data. The *MarLIN* programme operates an internal quality assurance procedure, to ensure that only the most accurate available information is provided on-line. The quality of the available evidence and our confidence in our assessments (based on availability of information) is clearly stated.

Table A2.7. Species that indicate biotope sensitivity.

Rank	Criteria
SELECTION CRITERIA The following criteria are used to decide which species best represent the sensitivity of a biotope or community as a whole.	
Rank	**Criteria**
Key structural species	The species provides a distinct habitat that supports an associated community. Loss/degradation of the population of this species would result in loss/degradation of the biotope.
Key functional species	The species maintains community structure and function through interactions with other members of that community (for example, predation, grazing, and competition). Loss/degradation of the population of this species would result in rapid, cascading changes in the biotope.
Important characterizing species	The species is/are characteristic of the biotope and are important for the classification of the biotope. Loss/degradation of the population of these species would result in loss of that biotope.
Important structural species	The species positively interact with the key or characterizing species and is important for their viability. Loss/degradation of populations of these species would result likely reduce the viability of the key or characterizing species. For example, these species may prey on parasites, epiphytes, or disease organisms of the key or characterizing species.
Important functional	The species is/are the dominant source of organic matter or primary production within the ecosystem. Loss/ degradation of these species could result in changes in the community function and structure.
Important other species	Additional species that do not fall under the above criteria but where present knowledge of the ecology of the community suggests they may affect the sensitivity of the community.
Note: All key species will be used in the sensitivity assessment. However, where several important species satisfy the above criteria examples from each rank should be used. Preference should be given to examples where direct evidence of community interaction is available or they are characteristic (highly faithful) of the biotope.	

5. Assess the intolerance, recoverability, and sensitivity of indicative species to environmental factors. The sensitivity of the indicative species is assessed with respect to change in 24 separate environmental factors (see Table A2.3 above). Precedence is given to direct evidence of effect or impact. In the absence of direct evidence, the *MarLIN* rationale includes simple decision trees to aid intolerance and recoverability assessment based on the available information. The decision trees provide a systematic and transparent approach to assessment. The decision trees are described in full by Tyler-Walters *et al.* (2001).

6. Assess overall intolerance and recoverability of the biotope. The intolerance and recoverability of the biotope are derived from the intolerance and recoverability of the species identified as indicative of sensitivity, using a simple procedure shown in Figure A2.1 for intolerance and in Figure A2.2 for recoverability. The definitions of biotope intolerance (revised in April 2003) are shown in Table A2.8.

Knowledge of the biology of other species in the biotope, especially if they have been researched as a part of the *MarLIN* programme, is also taken into account.

Precedence is given to direct evidence of the effects of changes in environmental factors on a habitat, its community and associated species (i.e. the components of a biotope), and its subsequent recovery. The intolerance of a biotope to change in each environmental factor is assessed against a standard 'benchmark' level of effect, which allows the user to compare the recorded sensitivity to the level of effect predicted to be caused by a proposed development or activity. The evidence and key information used to assess intolerance, recoverability, and sensitivity, and any judgements made are explained in the on-line rationale for each assessment. The source of all information used is clearly referenced on-line.

Table A2.8. Biotope intolerance (previously 'sensitivity' and revised April 2003)

BIOTOPE INTOLERANCE The susceptibility of a habitat, community or species (i.e. the components of a biotope) to damage, or death, from an external factor. Intolerance must be assessed relative to change in a specific factor.	
Rank	**Definition**
High	Species important for the structure and/or function of the biotope, or its identification ('important characterizing' species), are likely to be killed and/or the habitat is likely to be destroyed by the factor under consideration.
Intermediate	The population(s) of species important for the structure and/or function of the biotope, or its identification ('important characterizing' species), may be reduced or 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	Species important for the structure and/or function of the biotope, or its identification ('important characterizing' species), will not be killed or destroyed by the factor under consideration and the habitat is unlikely to be damaged. However, the viability of a species population or the diversity / functionality in a community will be reduced.
Tolerant	The factor does not have a detectable effect on the structure and/or function of a biotope or the survival or viability of species important for the structure and/or function of the biotope or its identification.
Tolerant*	The extent or species richness of a biotope may be increased or enhanced by the factor.
Not relevant	Intolerance 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).

7. Assess sensitivity of the biotope. The overall sensitivity rank is derived from the combination of intolerance and recoverability using the rationale shown in Tables A2.5 and A2.6 above.

8. Assess the likely effect of the environmental factors on species richness. Change in an environmental factor may not significantly damage key or important species but may still degrade the integrity of the biotope due to loss of species richness. Therefore, the likely effect of the factor on species richness in the biotope is indicated (see Table A2.9).

9. Signing-off. *MarLIN* reviews are checked by the Programme Director for accuracy and clarity and the required changes made before the review goes 'on-line' on the Web site.

10. Referee. As a final stage in the *MarLIN* quality assurance, Key Information reviews are subject to peer review by an external marine biologist where possible.

Table A2.9. The likely response of species richness to an external factor

SPECIES RICHNESS The number of species in a given habitat, biotope, community or assemblage	
The following scale is used to judge the likely response of species richness to an external factor.	
Rank	**Definition**
Major decline	The number of species in the community is likely to decrease significantly (>75% of species) in response to the factor, probably because of mortality and loss of habitat. For example, a change from very rich to very poor on the NHAP scales (Hiscock, 1996).
Decline	The community is likely to loose some of its species in response to the factor by either direct mortality or emigration.
Minor decline	The community is likely to loose few species (<25% of species) in response to the factor. For example, a decrease of one level on the NHAP scales (Hiscock 1996).
No change	The factor is unlikely to change the species richness of the community
Rise	The number of species in the community may increase in response to the factor. (Note the invasion of the community by aggressive or non-native species may degrade the community).
Not relevant	It is extremely unlikely for a factor to occur (e.g. emergence of a deep water community) or the community is protected from the factor.

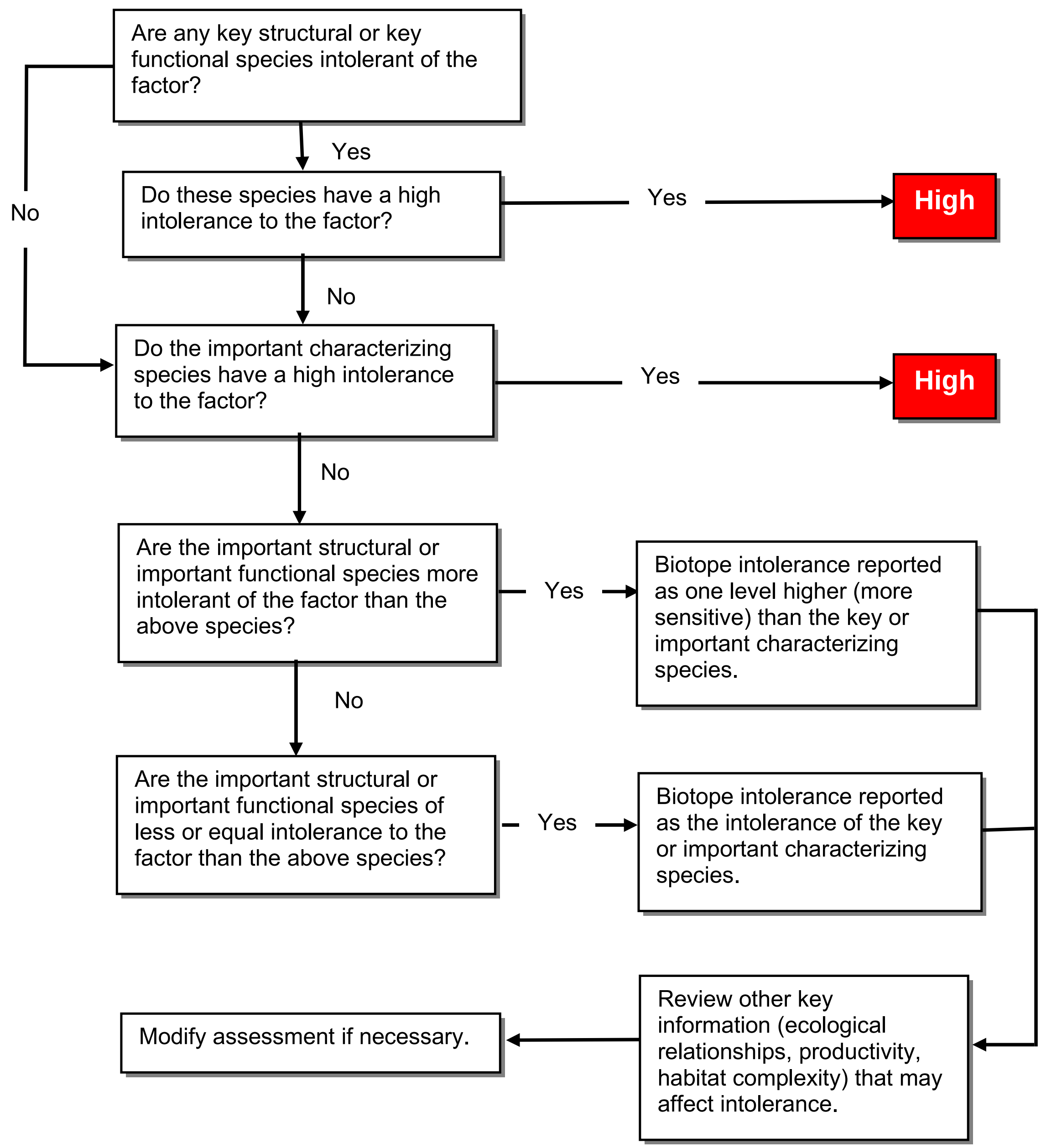

Figure A2.1. Biotope 'intolerance' assessment rationale.

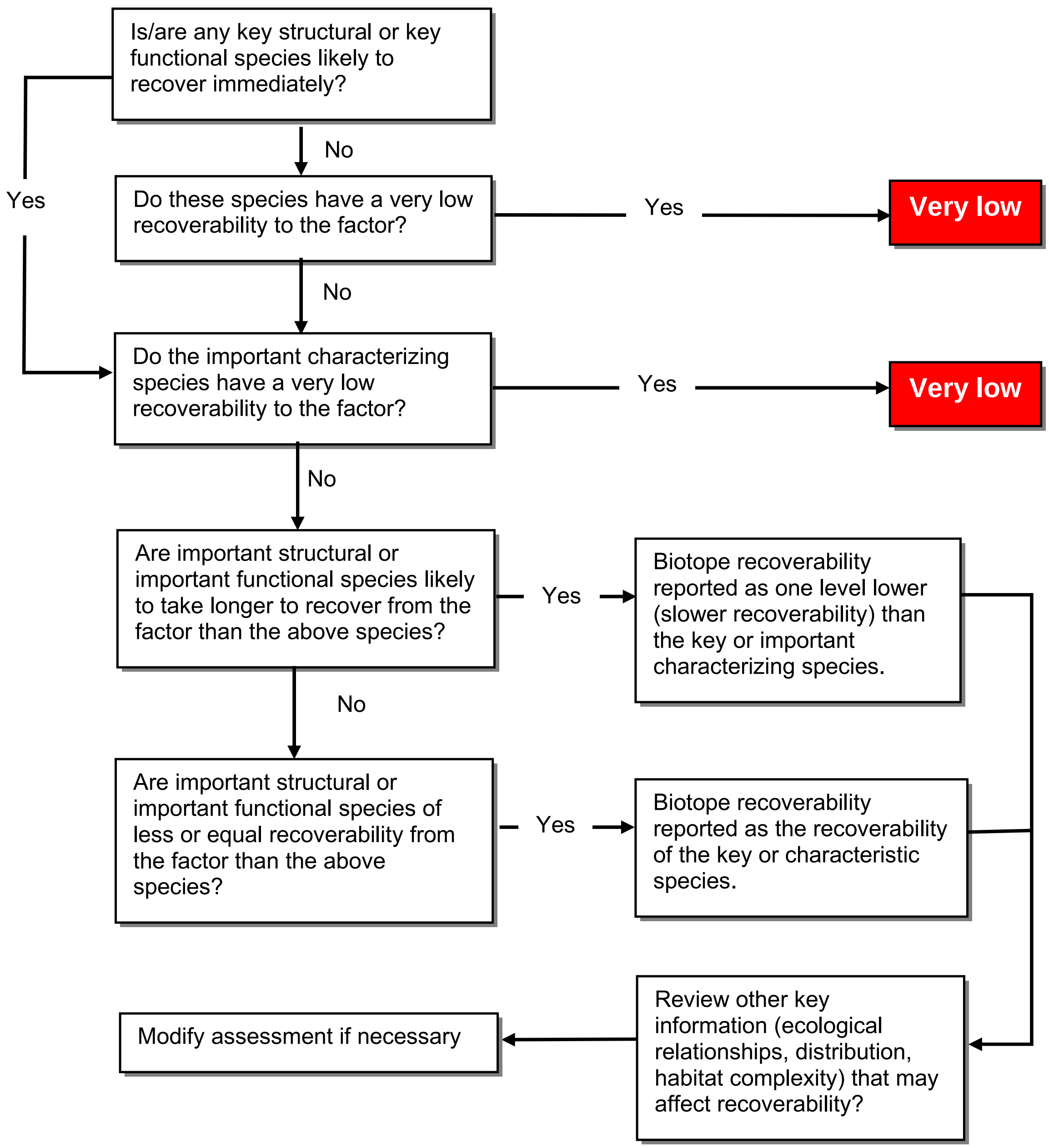

Figure A2.2. Biotope 'recoverability' assessment rationale.

Appendix 3. Guide to the interpretation of sensitivity assessments and the benchmarks used.

The following is a short summary of the key assumptions involved in *MarLIN* sensitivity assessments and notes on their interpretation. Information on the development of the sensitivity assessment approach is detailed in Hiscock *et al.* (1999) and Tyler-Walters & Jackson (1999), the full approach is outlined in Tyler-Walters *et al.* (2001) and, as revised in 2003, on the *MarLIN* Web site.

Introduction

Marine organisms may be affected by a number of human activities and natural events. The magnitude or scale of the effect of an activity (or event) is dependent on the receiving environment. The same activity (or event) in different locations may have different effects. For example, an activity that markedly increased siltation may have little effect in a turbid estuary whereas it would probably have significant effects in a sheltered embayment. Therefore, the effects of an activity and the resultant change in environmental factors are site specific and cannot be generalized.

In addition, any one activity (or event) may change one or more environmental factors (see 'effects of specified marine and coastal activities or natural events'). Similarly, it is not possible to take into account every set of environmental conditions to which a species or biotope are exposed throughout their range.

In order to achieve a practical, systematic, and transparent approach, the assessment of intolerance, recoverability, and sensitivity required a standard set of definitions and scales (see Tyler-Walters *et al.,* 2001, *MarLIN*, 2004). The assessment of intolerance required a specified level of environmental perturbation. Therefore, the *MarLIN* programme developed a set of 'benchmark' levels of environmental change in the environmental factors against which to assess sensitivity. The benchmarks also allow intolerance and hence sensitivity to be compared against the predicted effects of planned projects or proposals (see (see Tyler-Walters *et al.,* 2001, *MarLIN*, 2004).

Sensitivity assessments

Sensitivity assessments and key information reviews are designed to provide the information required to make scientifically based environmental management decisions. It is not possible for sensitivity assessments to consider every possible outcome and are indicative. *MarLIN* sensitivity assessments are indicative qualitative judgements based on the best available scientific information. *They do not allow quantitative analysis.* The sensitivity assessments represent the most likely (or probable) result of a given change in an environmental factor on a species population or biotope.

Sensitivity assessments require expert interpretation on a site-by-site or activity-by activity basis. *MarLIN* sensitivity assessments should be read in conjunction with the explanation and key information provided, together with the relevant benchmark. In all cases, an explanation of each intolerance, recoverability and hence sensitivity assessment is provided, together with a summary of the relevant key information, and references highlighted.

Assumptions

The following decisions and assumptions are inherent in the *MarLIN* approach to sensitivity assessment.

- The intolerance, recoverability, and sensitivity of a species or biotope to a specified level of environmental perturbation are dependent on the biology of the species or ecology of the biotope.

- Intolerance, and hence sensitivity, depends on the magnitude, duration, or frequency of change in a specific environmental factor.
- The effects of an activity or natural event and the resultant change in environmental factors are site specific and cannot be generalized. Therefore, a series of standard level of effect or change in each environmental factor are used for assessment (the benchmarks).
- *MarLIN* sensitivity assessments are not site specific. The intolerance of a hypothetical 'average' species population is assessed, representing a population in the middle of its range or habitat preferences. Populations at the limits of their environmental preferences are likely to be more intolerant of environmental perturbation.
- **Recoverability assumes that the impacting factor has been removed or stopped** and the habitat returned to a state capable of supporting the species or biotope in question. **The time taken for the habitat to return to a state capable of supporting the species or biotope is not assessed.**
- Where the collated key information and other evidence suggest a range of intolerances or recoverabilities, a precautionary approach is taken, and the 'worst case' scenario, i.e. the higher sensitivity, is reported.
- In all cases, the explanation behind each sensitivity assessment, the relevant key information and references are highlighted.

Interpretation of sensitivity assessments

Sensitivity is based on the assessment of intolerance against a benchmark level of change in an environmental factor, and the likely recoverability of the species population or biotope.

- The benchmarks are intended to be pragmatic guidance values for sensitivity assessment, to allow comparison of sensitivities between species, and to allow comparison with the predicted effects of project proposals.
- Species or biotopes are likely to be more intolerant, and hence potentially more sensitive, to any activity or natural event that causes a change in a specific environmental factor of greater magnitude and/or longer duration and/or greater frequency than the benchmark. For example:
    - if the predicted change in an environmental factor has a greater magnitude than that used in the benchmark, then it is likely that the species population / biotope will have a greater sensitivity to this change;
    - if the predicted change in an environmental factor has a longer duration than that used in the benchmark, then it is likely that the species population / biotope will have a greater sensitivity to this change;
    - if the predicted change in an environmental factor is likely to occur at higher frequency than used in the benchmark, then it is also likely that the species or community will exhibit a higher sensitivity;
    - if the frequency of the predicted change in an environmental factor is greater than the time required for recover then the species or community will probably exhibit a higher sensitivity,
    - while if the species or community is likely to recover between the impacting events then it may not exhibit an increased sensitivity.
- Similarly, if a species population is isolated from sources of recruitment, for instance in isolated water bodies (e.g. sea lochs or lagoons) or by hydrography, then the

recoverability may lower, and hence the population may exhibit a higher sensitivity. Isolation is already factored into the recoverability assessments for relevant biotopes and lagoonal species.

Activities that result in incremental long term change, such as climate change, are difficult to assess since the given level of change varies with time. Synergistic and antagonistic effects are also difficult to predict and are poorly understood, especially for pollutants. **These effects have not been addressed within the sensitivity assessments.** However, benchmarks could be compared to the predicted level of change at specific time intervals.

Appendix 4. Sensitivities of researched biotopes in the vicinity of Coasteering activities in west Wales

Biotope name	Biotope code	Intolerance	Recovery	Sensitivity	Explanation	Confidence
Yellow and grey lichens on supralittoral rock	LR.YG	Intermediate	Low	High	The biotope is susceptible to trampling by birds (in heavily populated sites) animals and man, animal rubbing and the physical abrasion caused by wind. Physical abrasion by wind may discourage large foliose and fructicose lichens. Fletcher (1980) noted that large specimens of lichens, e.g. *Ramalina siliquosa*, were only found on vertical rocks inaccessible to animals, including man. James & Syratt (1987) noted that rubbing by animals (e.g. sheep) damaged lichens resulting in loss of parts of some thalli and loss of *Ramalina siliquosa* at some sites, while it showed signs of regeneration at some sites. Trampling damage is greatest when the thallus is wet, causing it too peel from the surface, while when dry, some fragments are likely to remain to propagate the lichen (Fletcher, 1980). Physical disturbance of the lichen flora or substratum may reduce species richness and favour more rapid growing, disturbance tolerant species, e.g. *Lecanora dispersa*, *Candelariella vitellina* and *Rinodina gennerii* (Fletcher, 1980). Extreme physical abrasion due to high pressure water cleaning techniques (used to clear oil after spills), damaged lichens even at low pressures, especially *Ramalina siliquosa*, *Xanthoria* sp. and *Caloplaca marina*, and removed all supralittoral lichens at high pressures (Crump & Moore, 1997; Menot *et al.*, 1998). Overall, supralittoral lichens appear to be intolerant of physical abrasion. Animal trampling and rubbing are likely to remove a proportion of lichen thalli in the short term, and alter the lichen communities in the long term. Therefore, an intolerance of intermediate has been recorded. Recovery is likely to be slow (see Appendix 5).	Moderate

Biotope name	Biotope code	Intolerance	Recovery	Sensitivity	Explanation	Confidence
Chrysophyceae on vertical upper littoral fringe soft rock	LR.Chr	Intermediate	Very high	Low	The 'Chrysophyceae' mat is very thin (a few millimetres) and the *Pseudendoclonium submarinum* belt exists as a thin coasting of the rock. These algal communities are likely to be removed as a result of any abrasion, e.g. from stranding or trampling, especially where the friable rock surface is removed. Therefore, an intolerance of intermediate has been recorded. However, recovery is likely to be rapid if suitable substratum remains (see Appendix 5).	Moderate
Mytilus edulis and barnacles on very exposed eulittoral rock	ELR.MytB	Intermediate	High	Low	Physical disturbance due to wave driven debris (e.g. logs) or sediment abrasion will remove patches of mussels (see Daly & Mathieson, 1977, Seed & Suchanek, 1992) and probably areas of barnacles. Rocky shore communities may also suffer physical disturbance due to trampling. Brosnan & Cumrine (1994) reported that beds of mussels were intolerant of trampling, depending on the thickness and density of the bed. In areas of intense trampling, mussels were uncommon and restricted to crevices. Trampling also reduced recruitment, and hence recovery rates (Brosnan & Cumrine, 1994). Small patches of mussels may be more susceptible to trampling damage. Brosnan & Cumrine (1994) also reported that barnacles were crushed by trampling and that *Mastocarpus stellatus* was intolerance of moderate trampling. Coralline algal turf was adversely affected by trampling, which resulted in a 50% reduction in turf height in one study (Brown & Taylor, 1999; Schiel & Taylor, 1999). Schiel & Taylor (1999) and Fletcher & Frid (1996) reported that trampling resulted in loss of species richness and formation of bare space in intertidal macroalgal assemblages. Amphipods, gastropods or small limpets may be crushed by trampling. Therefore, a proportion of the population or mussels and barnacles may be lost, together with members of the associated community and macroalgae due to trampling (see species reviews for details) and an intolerance of intermediate has been recorded. Recovery is probably high (see Appendix 5). Severe abrasion (e.g. the barge	Moderate

Biotope name	Biotope code	Intolerance	Recovery	Sensitivity	Explanation	Confidence
					stranding described by Bennell, 1981) will be similar to substratum removal in effect (see above).	
Barnacles and *Patella* spp. on exposed or moderately exposed, or vertical sheltered, eulittoral rock	ELR.BPat	Intermediate	High	Low	The biotope is susceptible to abrasion and physical disturbance from trampling. Light trampling pressure, of 250 steps in a 20x20 cm plot, one day a month for a period of a year, has been shown to damage and remove barnacles (Brosnan & Crumrine, 1994). Trampling pressure can thus result in an increase in the area of bare rock on the shore (Hill *et al.*, 1998). Chronic trampling can affect community structure with shores becoming dominated by algal turf or crusts. Therefore, chronic trampling could result in loss of the biotope and an intolerance of high has been recorded. However, if trampling stops, recovery should be good. In Oregon for example, the algal-barnacle community recovered within a year after trampling stopped (Brosnan & Crumrine, 1994). Bennell (1981) observed the impact of abrasion of barnacles from a grounded barge at Amlwch, North Wales and subsequent recovery of barnacle populations at least within three years.	High
Corallina officinalis on very exposed lower eulittoral rock	ELR.Coff	Intermediate	Very high	Low	Abrasion by an anchor or mooring may remove some fronds of the foliose red algae and coralline turf, although most species would grow back from their remaining holdfasts. Trampling may be more damaging. For example, moderate (50 steps per 0.09 sq. metre) or more trampling on intertidal articulated coralline algal turf in New Zealand reduced turf height by up to 50, and the weight of sand trapped within the turf to about one third of controls. This resulted in declines in densities of the meiofaunal community within two days of trampling. Although the community returned to normal levels within 3 months of trampling events, it was suggested that the turf would take longer to recover its previous cover (Brown & Taylor 1999). Similarly, Schiel & Taylor (1999) noted that trampling had a direct detrimental effect on coralline turf species on the New Zealand rocky shore. At one site, coralline bases were seen to peel from the rocks (Schiel & Taylor 1999), although this was probably due to increased	Moderate

Biotope name	Biotope code	Intolerance	Recovery	Sensitivity	Explanation	Confidence
					desiccation caused by loss of the algal canopy. The crustose base has nearly twice the mechanical resistance (measured by penetration) of fronds (Littler & Kauker, 1984). Brosnan & Cumrine (1994) also reported that foliose algae, e.g. *Mastocarpus papillatus* showed significant declines in cover in response to trampling, although recovery was rapid, probably from remaining holdfasts. Therefore, physical abrasion due to trampling is likely to result in a significant decline in the cover of the coralline turf, red algae and epiphytic fauna and an intolerance of intermediate has been recorded. The dominant algae are likely to recover rapidly by regrowth from remaining fronds and holdfasts, and epiphytic fauna will colonize the turf relatively quickly.	
Himanthalia elongata and red seaweeds on exposed lower eulittoral rock	ELR.Him	Low	High	Low	Abrasion may damage fronds of established seaweed and crush germlings and faunal species. However, *Patella vulgata*, for instance has a tough shell which offers protection from any abrading factors and any near vibration causes the shell muscles to contract vigorously, clamping the animal to the rock. However, a short, sharp knock may dislodge an individual leaving it vulnerable to predation. In the eulittoral zone, abrasion caused by human trampling has been shown to reduce algal cover on shores (Holt *et al.*, 1997), and may be of more relevance to the biotope than the dropping and dragging of an anchor as an abrasive factor. Brosnan & Crumrine (1994) found that the foliose red algae *Mastocarpus papillatus* was intolerant of moderate levels of trampling. However, trampling pressure may not be particularly intense either considering that the biotope occurs within the vicinity of the low water mark and therefore may be inaccessible for much of the time. At the benchmark level intolerance has been assessed to be low and recovery high as a population would remain *in situ* (see Appendix 5).	Moderate

Biotope name	Biotope code	Intolerance	Recovery	Sensitivity	Explanation	Confidence
Barnacles and fucoids (moderately exposed shores)	MLR.BF	High	High	Moderate	The rocky intertidal is not at risk from boating activity but is susceptible to abrasion and physical impact from trampling. Even very light trampling on shores in the north east of England was sufficient to reduce the abundance of fucoids (Fletcher & Frid, 1996) which, in turn reduced the microhabitat available for epiphytic species. Trampling damage is particularly serious for the long-lived but slowly recruiting *Ascophyllum nodosum*. Light trampling pressure, of 250 steps in a 20x20cm plot, one day a month for a period of a year, has also been shown to damage and remove barnacles (Brosnan & Crumrine, 1994). Trampling pressure can thus result in an increase in the area of bare rock on the shore (Hill *et al.*, 1998). Chronic trampling can affect community structure with shores becoming dominated by algal turf or crusts. However, if trampling stops, recovery should be good. In Oregon for example, the algal-barnacle community recovered within a year after trampling stopped (Brosnan & Crumrine, 1994).	Moderate
Underboulder communities	MLR.Fser .Fser.Bo	High	High	Moderate	In addition to disturbance caused by wave energy, intertidal boulder communities are often disturbed by, for example, bait collectors, inquisitive school groups and field researchers. Boulders left overturned place the organisms on the now upward facing part of the boulder at great risk of desiccation (see Desiccation above). Furthermore, many stable boulders are fused together by algal growth (especially corallines) and breaking this matrix would be very harmful (Foster-Smith, pers. comm.). Furthermore, this disturbance and habitat degradation could change a stable boulder field to an unstable field on a long-term basis (Foster-Smith, pers. comm.). Movement of the boulder surface against other hard surfaces (for instance, during extreme storm events) is likely to cause significant damage to encrusting fauna that is characteristic of the community. Recoverability is expected to be high (see additional information).	Moderate

Biotope name	Biotope code	Intolerance	Recovery	Sensitivity	Explanation	Confidence
Ulva spp. on freshwater-influenced or unstable upper eulittoral rock	MLR.Ent	High	Very high	Low	*Ulva intestinalis* and *Porphyra* species are likely to be susceptible to abrasion as they are not of a resilient growth form and would easily be scraped from the substratum by dragging objects. Intolerance has been assessed to be high. However, both are cosmopolitan species that reproduce rapidly enabling them to colonize available substrata, so recoverability has been assessed to be very high (see additional information, below).	Moderate
Rhodothamniella floridula on sand-scoured lower eulittoral rock	MLR.Rho	Intermediate	High	Low	Although algal species are highly flexible, abrasion is likely to cause damage to and removal of fronds and may even remove entire plants from the substratum. The cushion-like base of turf forming algae (such as *Rhodothamniella floridula*) may offer some protection against abrasion but if a portion is removed, the sharp edges may be subject to lifting by wave action. Limpets, *Mytilus edulis* and barnacle populations have also been reported to be dislodged by abrasion or physical disturbance. Intolerance has been assessed to be intermediate. Recoverability is likely to be high (see Appendix 5).	Moderate
Mytilus edulis and *Fucus vesiculosus* on moderately exposed mid eulittoral rock	MLR.MytFves	Intermediate	High	Low	Daly & Mathieson (1977) reported that the lower limit of *Mytilus edulis* populations at Bound Rock, USA, was determined by burial or abrasion by shifting sands. Wave driven logs have been reported to influence *Mytilus edulis* populations, causing the removal of patches from extensive beds that subsequently open the beds to further damage by wave action. It is likely that abrasion or impact at the level of the benchmark would also damage or remove patches of the population. The effects of trampling on *Mytilus californianus* beds in Australia were studied by Brosnan & Cumrine (1994). They concluded that mussel beds were intolerant of trampling, depending on bed thickness, and noted that in heavily tramped site mussels were uncommon and restricted to crevices. Trampling also inhibited subsequent recovery. Trampling pressure was most intense in spring and summer, so that gaps and patches created by storms in winter were not repaired but exacerbated.	Moderate

Biotope name	**Biotope code**	**Intolerance**	**Recovery**	**Sensitivity**	**Explanation**	**Confidence**
					Fucoid cover has also been reported to be reduced by trampling (Holt *et al.*, 1997). Brosnan & Cumrine (1994) also observed that barnacles were crushed and removed by trampling in California but recovery took place within one year following the cessation of trampling. Therefore, it is likely that abrasion and physical disturbance at the benchmark level will result in loss of a proportion of the mussel patches, fucoids and their associated species and an intolerance of intermediate has been recorded. Recoverability is likely to be high (see Appendix 5). Large scale abrasion e.g. due to a vessel grounding, is likely to be similar to substratum loss in effect.	
Sabellaria alveolata reefs on sand-abraded eulittoral rock	MLR.Salv	Intermediate	High	Low	Cunningham *et al.* (1984) examined the effects of trampling on *Sabellaria alveolata* reefs. The reef recovered within 23 days from the effects of trampling, (i.e. treading, walking or stamping on the reef structures) repairing minor damage to the worm tube porches. However severe damage, estimated by kicking and jumping on the reef structure, resulted in large cracks between the tubes, and removal of sections (ca 15x15x10 cm) of the structure (Cunningham *et al.*, 1984). Subsequent wave action enlarged the holes or cracks. However, after 23 days, at one site, one side of the hole had begun to repair, and tubes had begun to extend into the eroded area. At another site, a smaller section (10x10x10 cm) was lost but after 23 days the space was already smaller due to rapid growth. Cunningham *et al.* (1984) reported that *Sabellaria alveolata* reefs were more tolerant of trampling than expected but noted that cracks could leave the reef susceptible to erosion and lead to large sections of the reef being washed away. However, eroded sections can survive and may lead to colonization of previously unsettled areas. The strange sculpturing of colonies in some areas is probably due to a combination of erosion and recovery (Cunningham *et al.*, 1984). Continuous trampling may be more detrimental. For example, Holt *et al.* (1998) reported that, in Brittany, damage to reefs on popular beaches was limited to gaps created by trampling through the reef.	Moderate

Biotope name	Biotope code	Intolerance	Recovery	Sensitivity	Explanation	Confidence
					Once gaps are formed, they may be enlarged by wave action. The main cause of colony destruction is through wave action. Cunningham *et al.* (1984) also noted that collection of *Sabellaria alveolata*, although a rare occurrence, may be particularly damaged as it will involve removal of sections of the reef. Trampling has been reported to reduce fucoid cover (Holt *et al.*, 1997). Similarly, littorinids will be probably displaced and very occasionally crushed by trampling, although at the population level the effects are probably minimal. Therefore, trampling and other physical disturbance can potentially remove a proportion of the reef and an intolerance of intermediate has been recorded. Variability in *Sabellaria alveolata* recruitment (dependent on suitable environmental conditions) means that recovery could take a several years. The presence of remaining adults will assist in larval settlement, as this is the preferred substratum (Wilson, 1929). Therefore recoverability has been assessed as high.	
Ascophyllum nodosum on very sheltered mid eulittoral rock.	SLR.Asc	High	Low	High	Trampling on the rocky shore has been observed to reduce fucoid cover which decreased the microhabitat available for epiphytic species, increased bare space and increased cover of opportunistic species such as *Ulva* (Fletcher & Frid, 1996). *Ascophyllum nodosum* seems to be particularly intolerant of damage from trampling (Flavell, 1995 cited in Holt *et al.*, 1997). It is also likely to be removed if shores are mechanically cleaned following oil spills. Light trampling pressure has also been shown to damage and remove barnacles (Brosnan & Crumrine, 1994). Thus, trampling can significantly affect community structure and intolerance has, therefore, been assessed as high. *Ascophyllum nodosum* has poor recruitment rates and is slow growing, limiting recovery (Holt *et al.*, 1997). The lack of recovery of *Ascophyllum nodosum* from harvesting is well documented. For example, in their work on fucoid recolonization of cleared areas at Port Erin, Knight and Parke (1950) observed that even eight years after the original clearance there was still no sign of the	High

Biotope name	**Biotope code**	**Intolerance**	**Recovery**	**Sensitivity**	**Explanation**	**Confidence**
					establishment of an *Ascophyllum nodosum* population. Therefore, recovery is likely to be low.	
Fucus ceranoides on reduced salinity eulittoral rock	SLR.Fcer	Intermediate	High	Low	Abrasive forces can damage and remove fronds and germlings of *Fucus ceranoides* and other algae. Abrasion caused by human trampling can significantly reduce the cover of fucoid algae on a shore (Holt *et al.*, 1997) and may be the most relevant source of abrasion and physical disturbance to the SLR.Fcer biotope. Therefore, intolerance has been assessed to be intermediate. Recoverability of fucoid species (except *Ascophyllum nodosum*) and faunal species is likely to be high (see Appendix 5).	Moderate
Barnacles and *Littorina littorea* on unstable eulittoral mixed substrata	SLR.BLlit	Intermediate	High	Low	The biotope is susceptible to abrasion and physical impact from trampling. Light trampling pressure, of 250 steps in a 20x20cm plot, one day a month for a period of a year, has been shown to damage and remove barnacles (Brosnan & Crumrine, 1994). Trampling pressure can thus result in an increase in the area of bare rock on the shore (Hill *et al.*, 1998). However, this biotope is characterized by unstable substrata, and probably experiences periodic episodes of severe abrasion from boulders turned or mobilized by wave action. Therefore, the species present are in all probability of at least intermediate intolerance to abrasion, a proportion of their populations being removed by abrasion. However, recovery is likely to be high (see Appendix 5) so that the biotope as a whole is probably of low sensitivity to abrasion.	Moderate
Fucus vesiculosus on mid eulittoral mixed substrata	SLR.FvesX	Intermediate	High	Low	Abrasion may damage fronds of established seaweed and crush germlings and faunal species. *Patella vulgata*, for instance has a tough shell that offers protection from any abrading factors and any near vibration causes the shell muscles to contract vigorously, clamping the animal to the rock. However, a short, sharp knock may dislodge an individual leaving it vulnerable to predation. In the intertidal zone, abrasion caused by human trampling may be the most relevant source of abrasion and physical disturbance in the biotope. Trampling has been shown to reduce algal cover on shores	Moderate

Biotope name	Biotope code	Intolerance	Recovery	Sensitivity	Explanation	Confidence
					(Holt *et al.*, 1997). At the benchmark level intolerance has been assessed to be intermediate as some plants would be removed and some animals crushed. Recoverability has been assessed to be high as a population would remain *in situ* (see Appendix 5).	
Green seaweeds (*Ulva* spp. and *Cladophora* spp.) in upper shore rockpools	LR.G	High	Very high	Low	*Ulva intestinalis* and *Cladophora rupestris* are likely to be susceptible to abrasion as they are not of a resilient growth form and would easily be scraped from the substratum by dragging objects. Littorinids may be knocked from rocks by physical disturbance and unless damaged are likely to reattach. Small copepods would probably be able to avoid abrasive agents by seeking refuge. Intolerance has been assessed to be high. *Ulva intestinalis* and *Cladophora rupestris* are cosmopolitan species that reproduce rapidly enabling them to colonize available substrata, so recoverability has been assessed to be very high (see Appendix 5).	Moderate
Corallina officinalis and coralline crusts in shallow eulittoral rockpools.	LR.Cor	Intermediate	High	Low	Abrasion by an anchor or mooring may remove some fronds of the foliose red algae and scrape coralline turf, although most species would grow back from their remaining holdfasts. Trampling may be more damaging (see ELR.Coff) but is likely to be less significant in rockpool than on open rock surfaces. Therefore an intolerance assessment of intermediate has been made.	Low
Fucoids and kelps in deep eulittoral rockpools	LR.FK	Intermediate	High	Low	Abrasion by an anchor or mooring may remove some fronds of the large macroalgae, foliose red algae and coralline turf, although most species would grow back from their remaining holdfasts. However, trampling may be more damaging. Deep pools are protected by their depth but shallower pools or the shallower margins of larger pools are probably more vulnerable. No studies of the effects of trampling on rockpools were found but studies of the effects on emergent algal communities are probably indicative. For example, moderate (50 steps per 0.09 square metre) or more trampling on intertidal articulated coralline algal turf in New Zealand reduced turf height by up to 50%, and the weight of sand trapped within the turf to about one third of controls. This resulted in	Low

Biotope name	Biotope code	Intolerance	Recovery	Sensitivity	Explanation	Confidence
					declines in densities of the meiofaunal community within two days of trampling. Although the community returned to normal levels within 3 months of trampling events, it was suggested that the turf would take longer to recover its previous cover (Brown & Taylor, 1999). Similarly, Schiel & Taylor (1999) noted that trampling had a direct detrimental effect on fucoid algae and coralline turf species on the New Zealand rocky shore. Low trampling intensity (10 tramples) reduced fucoid cover by 25%, while high intensity (200 tramples) reduced fucoid cover by over 90%, although over 97% cover returned within 21 months after spring trampling; autumn treatments took longer to recover due to the delay in recruitment. Coralline bases were seen to peel from the rocks (Schiel & Taylor, 1999) due to increased desiccation caused by loss of the algal canopy. Brosnan & Cumrine (1994) demonstrated that foliose species (e.g. fucoids and *Mastocarpus papillatus*) were the most susceptible to trampling disturbance, while turf forming species were more resistant. Barnacles were also crushed and removed. However, the algae and barnacles recovered in the year following the trampling (Brosnan & Cumrine, 1994). Similarly, Boalch *et al.* (1974) and Boalch & Jephson (1981) noted a reduction in fucoid cover (especially of *Ascophyllum nodosum*) at Wembury, Devon, when compared with the same transects surveyed 43 years previously. They suggested that the reduction in fucoid cover was due to the large number of visitors and school groups received by the site. Rockpools form natural mesocosms and so attract considerable attention from the general public, educational events and scientists alike. In addition to trampling within shallower pools and the margins of deeper pools, turning of rocks within the pool is likely to disturb underboulder communities (e.g. see MLR.Fser.Fser.Bo). Overall, a proportion of the macroalgal community and the invertebrates it supports are likely to be removed, depending on trampling intensity, and an intolerance of intermediate has been recorded. Recoverability is likely to be high (see Appendix 5) once trampling	

Biotope name	Biotope code	Intolerance	Recovery	Sensitivity	Explanation	Confidence
					has stopped. However, it should be noted that ongoing trampling is likely to result in a long term reduction in the diversity of the margins of the affected pools.	
Seaweeds in sediment (sand or gravel)-floored eulittoral rockpools	LR.SwSed	Intermediate	High	Low	Abrasion by an anchor or mooring may remove some fronds of the large macroalgae, foliose red algae and coralline turf, although most species would grow back from their remaining holdfasts. However, trampling and netting for shrimps or fish may be more damaging. Deep pools and the species they contain are protected by their depth but both small and large shallow pools are probably more vulnerable. No studies of the effects of trampling or netting on rockpools were found but studies of the effects on emergent algal communities are probably indicative. For example, moderate (50 steps per 0.09 sq. metre) or more trampling on intertidal articulated coralline algal turf in New Zealand reduced turf height by up to 50%, and the weight of sand trapped within the turf to about one third of controls. This resulted in declines in densities of the meiofaunal community within two days of trampling. Although the community returned to normal levels within 3 months of trampling events, it was suggested that the turf would take longer to recover its previous cover (Brown & Taylor, 1999). Similarly, Schiel & Taylor (1999) noted that trampling had a direct detrimental effect on fucoid algae and coralline turf species on the New Zealand rocky shore. Low trampling intensity (10 tramples) reduced fucoid cover by 25%, while high intensity (200 tramples) reduced fucoid cover by over 90%, although over 97% cover returned within 21 months after spring trampling; autumn treatments took longer to recover due to the delay in recruitment. Coralline bases were seen to peel from the rocks (Schiel & Taylor, 1999) due to increased desiccation caused by loss of the algal canopy. Brosnan & Cumrine (1994) demonstrated that foliose species (e.g. fucoids and *Mastocarpus papillatus*) were the most susceptible to trampling disturbance, while turf forming species were more	Low

Biotope name	Biotope code	Intolerance	Recovery	Sensitivity	Explanation	Confidence
					resistant. Barnacles were also crushed and removed. However, the algae and barnacles recovered in the year following the trampling (Brosnan & Cumrine, 1994). Boalch *et al.* (1974) and Boalch & Jephson (1981) noted a reduction in fucoid cover (especially of *Ascophyllum nodosum*) at Wembury, Devon, when compared with the same transects surveyed 43 years previously. They suggested that the reduction in fucoid cover was due to the large number of visitors and school groups received by the site. Dethier (1984) noted that low shore rockpools on the coast of Washington State, suffered physical disturbance from storms (wave action and wave driven logs) in winter months. The frequency of disturbance ranged from one every 2-5 years, while recovery of dominant to species to its original level ranged from 3 month to over 2 years. As a result, she estimated that ca 20-50% of the populations of dominant pools species were in a state of recovery in her study area. Rockpools form natural mesocosms and so attract considerable attention from the general public, educational events and scientists alike. In addition to trampling within shallow pools and the vicinity of deeper pools, turning of rocks within the pool is likely to disturb underboulder communities (e.g. see MLR.Fser.Fser.Bo). Overall, a proportion of the macroalgal community and the invertebrates it supports are likely to be removed, depending on trampling intensity, and an intolerance of intermediate has been recorded. Recoverability is likely to be high (see Appendix 5) once physical disturbance has stopped. However, it should be noted that ongoing trampling is likely to result in a long term reduction in the diversity of affected pools.	

Biotope name	Biotope code	Intolerance	Recovery	Sensitivity	Explanation	Confidence
Hydroids, ephemeral seaweeds and Littorina littorea in shallow eulittoral mixed substrata pools	LR.H	Intermediate	Very high	Low	Abrasion by an anchor or fishing gear could potentially destroy the biotope, depending on the size of the pool and on the size off the impact. The delicate filamentous fronds of *Ulva intestinalis* will easily be scraped off the surface of the rock. Parts of the delicate *Obelia longissima* colonies are also likely to be removed. However, the surface covering of hydrorhizae may remain largely intact, from which new uprights are likely to grow. In addition, the resultant fragments of colonies may be able to develop into new colonies (see displacement). If the shells of *Littorina littorea* or *Mytilus edulis* are damaged, the risk of predation and desiccation will increase. In most cases, it is likely that some part of each population will remain and therefore, intolerance has been assessed as intermediate. Recovery is likely to be very high (see additional information).	Low
Overhangs and caves	LR.Ov	High	High	Moderate	Communities under overhangs and in caves which extend to a seabed that is of mobile substrata demonstrate a zonation from bare (abraded) rock adjacent to the bottom, through fast settling and growing species to abrasion tolerant species to the typical overhang community (that is nevertheless probably subject to abrasion and damage during storms). Whilst whole communities are destroyed by abrasion from coarse sediments including especially pebbles and cobbles, recovery would occur from surviving colonies and individuals and new settlement from larval sources. Abrasion by human activities might include anchoring and dredging (fisheries).	High
Rhodothamniella floridula in upper littoral fringe soft rock caves	LR.RhoCv	Intermediate	High	Low	Upper littoral fringe caves are unlikely to be impacted by physical disturbance from anchorage or dredging. However, soft rocks are friable, and physical disturbance may be caused by pebbles, rocks, or marine debris, which accumulates in caves when moved by wave action. Although algal species are highly flexible, abrasion is likely to cause damage to and removal of fronds and may even remove entire plants from the substratum. The cushion-like base of turf forming algae (such as *Rhodothamniella floridula*) may offer some	Moderate

Biotope name	Biotope code	Intolerance	Recovery	Sensitivity	Explanation	Confidence
					protection against abrasion but if a portion is removed, the sharp edges may be subject to lifting by wave action. Intolerance has been assessed to be intermediate. Recoverability is likely to be high (see Appendix 5).	
Faunal crusts on wave-surged littoral cave walls	LR.FLR. CVOV.FaCr	Intermediate	High	Low	Due to the fact that this biotope is associated with cave habitats, abrasion and physical disturbance in this biotope is likely to come in the form of cobbles taken into suspension. Trampling and boats running aground are unlikely. Both the flora and fauna associated with this biotope are characterized by low lying crust forming species and therefore the effects of abrasion will most likely be the removal of, for example, small patches of sponge and bryozoan colonies or encrusting algae. Individual anemones may be killed but mass mortality is unlikely. In a study looking at the compressive strengths of several barnacles (Gubbay, 1983), *Semibalanus balanoides* was found to be weaker than *Balanus perforatus* and repeated physical disturbance in areas where these two co-existed could reduce the abundance of the weaker species thus altering the relative abundances of barnacles. In the lower reaches of the cave, suspended cobbles could scour the walls creating bare patches among the crusts and this area is likely to be more adversely affected than higher up the walls. Intolerance has been recorded as intermediate to reflect some mortality. Due to the fact that a proportion of each species will remain, recoverability is likely to be high.	Moderate
Alaria esculenta on exposed sublittoral fringe bedrock	EIR.Ala	Low	High	Low	Moderate trampling on articulated coralline algal turf in the New Zealand intertidal (Brown & Taylor 1999; Schiel & Taylor 1999) resulted in reduced turf height, declines in turf densities, and loss of crustose bases in some case probably due to loss of the canopy algae and resultant desiccation. Calcification is thought to an adaptation to grazing and sediment scour (Littler & Kauker 1984). The sublittoral fringe is unlikely to be significantly impacted by trampling due to its position of the lower shore but may be prone to	Low

Biotope name	Biotope code	Intolerance	Recovery	Sensitivity	Explanation	Confidence
					abrasion from moorings or low tide landings. Given its resilience to wave action *Alaria esculenta* is unlikely to be significantly damaged by abrasion although the under canopy coralline turf may suffer some damage. The coralline turf meiofauna will probably be lost as a result of trampling.	
Laminaria hyperborea forest with a faunal cushion (sponges and polyclinids) and foliose red seaweeds on very exposed upper infralittoral rock	EIR.LhypFa	Intermediate	Moderate	Moderate	Although tough enough to withstand strong wave action, species in this biotope are likely to be detached by abrasive force. For example, a passing scallop dredge would have an effect similar to kelp harvesting, although more localized and on a smaller scale (see below). Therefore an intolerance of intermediate has been recorded. Whilst recovery may occur for some species from parts remaining in crevices, others such as *Laminaria hyperborea* will have to recover from sporelings already present or that settle after the event. For recoverability, see Appendix 5.	Moderate
Laminaria hyperborea with dense foliose red seaweeds on exposed infralittoral rock.	EIR.LhypR	Intermediate	Moderate	Moderate	Laminarians and red algae are likely to be damaged abrasion due to anchor impact and sand or cobble scour. However, a passing scallop dredge is likely to remove or damage a proportion of the kelp and red algae present. Therefore, the community as a whole is likely to be of intermediate intolerance to physical disturbance at the benchmark level. This biotope will be more intolerant of higher levels or frequency of physical disturbance e.g. routine or numerous anchorages. For recoverability, see Appendix 5.	Moderate

Biotope name	Biotope code	Intolerance	Recovery	Sensitivity	Explanation	Confidence
Foliose red seaweeds on exposed or moderately exposed lower infralittoral rock	EIR.FoR	Intermediate	High	Low	The growth form of *Delesseria sanguinea* and other foliose red algae suggests that its lamina would probably be damaged by abrasion but not removed. However, a passing scallop dredge would probably tear off a large proportion of the macroalgae and remove any associated species with them. Mobile species, such as isopods would probably avoid damage. Similarly, individuals in crevices or overhangs would probably be unaffected. Bradshaw *et al.* (2000) suggested that fragile species such as urchins (e.g. *Echinus esculentus*), suffered badly from impact with a passing scallop dredge. Other sessile faunal species such as *Clavelina lepadiformis* have relatively delicate growth forms and are likely to be damaged or removed by a passing dredge. Intolerance has been assessed to be intermediate as populations of species may be partially destroyed but the biotope would still be recognized. Recoverability has been assessed to be high (see Appendix 5).	Low
Sponge crusts and anemones on wave-surged vertical infralittoral rock	EIR.SCAn	High	High	Moderate	Organisms living in the biotope are likely to be damaged or removed by physical disturbance. A few will escape as they are present in crevices or fissures. Overall, a high intolerance is expected. For recoverability, see Appendix 5.	High
Laminaria digitata on moderately exposed sublittoral fringe rock	MIR.Ldig.Ldig	Intermediate	High	Low	The fronds of *Laminaria digitata* are leathery and the whole plant is very flexible so a force equivalent to a scallop dredge or an anchor landing on or being dragged across the seabed, is unlikely to cause significant damage to the kelp bed as a whole. However, some plants may be fatally damaged or ripped off the substratum. Other algae and sessile species such as sponges and large solitary tunicates are likely to be especially intolerant of physical disturbance and so the biotope has been assessed as having intermediate intolerance.	Moderate

Biotope name	Biotope code	Intolerance	Recovery	Sensitivity	Explanation	Confidence
Laminaria digitata and piddocks on sublittoral fringe soft rock	MIR.Ldig.Pid	Intermediate	High	Low	The fronds of *Laminaria digitata* are leathery and the whole plant is very flexible so physical disturbance by a scallop dredge or an anchor landing on or being dragged across the seabed, is unlikely to cause significant damage to the kelp bed as a whole. However, some plants may be fatally damaged or ripped off the substratum. Other algae and sessile species such as sponges and large solitary tunicates are likely to be sensitive to abrasion and so the biotope as a whole has been assessed as having intermediate intolerance.	Low
Laminaria saccharina, *Chorda filum* and dense red seaweeds on shallow unstable infralittoral boulders or cobbles	MIR.Lsac ChoR	Intermediate	High	Low	This is a biotope that exists because of physical disturbance of mobile substrata. The community is likely to be destroyed by severe storms but will regenerate the following spring when conditions of wave action usually settle down. It might be that the biotope develops in a largely undisturbed way until the next sever storm, perhaps after several years. If disturbance occurs 'out-of-season', the biotope will be adversely affected for the remainder of the year.	Moderate
Alcyonium digitatum with a bryozoan, hydroid and ascidian turf on moderately exposed vertical infralittoral rock	IR.AlcByH	High	High	Moderate	Erect epifaunal species are particularly vulnerable to physical disturbance. Hydroids and bryozoans are likely to be removed or damaged by bottom trawling or dredging (Holt *et al.*, 1995). Veale *et al.* (2000) reported that the abundance, biomass and production of epifaunal assemblages decreased with increasing fishing effort. Hydroid and bryozoan matrices were reported to be greatly reduced in fished areas (Jennings & Kaiser, 1998 and references therein). The removal of rocks or boulders to which species are attached by the passage of mobile fishing gears (Bullimore, 1985; Jennings & Kaiser, 1998) results in substratum loss (see above). Magorrian & Service (1998) reported that queen scallop trawling removed emergent epifauna from horse mussel beds in Strangford Lough. They suggested that the emergent epifauna such as *Alcyonium digitatum* were more sensitive than the horse mussels themselves and reflected early signs of damage. However, *Alcyonium digitatum* is more abundant on high fishing effort grounds suggests that this seemingly fragile species is more resistant to abrasive disturbance than might be assumed (Bradshaw *et al.*, 2000), presumably owing	High

Biotope name	Biotope code	Intolerance	Recovery	Sensitivity	Explanation	Confidence
					to good recovery due to its ability to replace senescent cells, regenerate of damaged tissue and early larval colonization of available substrata. Epifaunal ascidians are also likely to be removed by physical disturbance. Overall, physical disturbance by mobile fishing gear or equivalent force is likely to remove a proportion of all groups within the community and attract scavengers to the community in the short term. Therefore, an intolerance of high has been recorded. Recoverability is likely to be high due to repair and regrowth of hydroids and bryozoans and recruitment within the community from surviving colonies and individuals (see Appendix 5). Severe physical disturbance will be similar in effect to substratum loss (see above).	

Appendix 5. Recovery potential of researched biotopes in the vicinity of Coasteering activities in west Wales

Biotope name	Recoverability
Yellow and grey lichens on supralittoral rock (LR.YG)	Sexual spores and asexual propagules of lichens are probably widely dispersed by the wind and mobile invertebrates, while the microalgal symbionts are probably ubiquitous. The thallus of lichens often dies from the centre out, growth occurring at the margin edge. However, fragments of the remaining thallus continue to grow, often faster than in a large thallus. Nevertheless, lichen growth rates are low, rarely more than 0.5-1mm/year in crustose species while foliose species may grow up to 2-5mm/year. Cullinane *et al.* (1975) noted that many of the lichens lost due to an oil spill in Bantry Bay were probably 20-50 years old based on their size, and life spans of lichens have been estimated to be 100 years or more (Jones *et al.*, 1974). Given their slow growth rates lichens will probably take many years to recover their original cover, possibly taking up to 20 years. Fletcher (1980) suggested that newly exposed substratum needs to be modified by weathering and that initiation of new thallus is thought to take several years. Crump & Moore (1997) observed that lichens had not colonized experimentally cleared substrata within 12 months. Brown (1974) reported that recolonization of substrata within Caerthillian Cove, Cornwall, which was heavily affected by oil and dispersants after the *Torrey Canyon* oil spill, took 7 years to begin. Overall, although mobile invertebrate fauna will probably recolonize rapidly, recovery of lichens communities from damage will probably take many years. In heavily damaged areas, the prolonged recolonization period and subsequent slow growth is likely to take a very long time and recovery rates are likely to be extremely slow, probably in excess of ten years (Holt *et al.*, 1995).
Chrysophyceae on vertical upper littoral fringe soft rock (LR.Chr)	Recovery will depend on regrowth from existing thallus or filaments and should be rapid. The 'Chrysophyceae' communities develop over winter, and several members of soft rock algal communities develop rapidly in spring. Most members of these algal communities produce motile spores or have a motile pelagic stage in their life cycle (except Cyanobacteria) and have the potential to disperse widely with effectively high fecundity. Therefore, once suitable habitat or environmental conditions return recovery is likely to be rapid, probably taking a year at most. However, little direct evidence of recovery rates was found.
Mytilus edulis and barnacles on very exposed eulittoral rock (ELR.MytB)	*Mytilus edulis* is highly fecund but larval mortality is high. Larval development occurs within the plankton over ca 1 month (or more), with high dispersal potential. Patches of mussels on the high shore and population of juveniles on filamentous substrata (e.g. macroalgae) probably contribute to local recruitment (Holt *et al.*, 1998). Larval supply and settlement could potentially occur annually. However, settlement is sporadic with unpredictable pulses of recruitment (Lutz & Kennish, 1992; Seed & Suchanek, 1992). Once settled, *Mytilus edulis* can reproduce within its first year if growth conditions allow. On rocky shores, gaps in mussel beds are often colonized by barnacles and fucoids, barnacles enhancing subsequent recruitment of mussels. Cycles of loss and recruitment lead to a patchy distribution of mussels on rocky shores. High intertidal and less exposed sites recovered slower than low shore, more exposed sites. Several long term studies showed that gaps took a long time to heal, but in some cases enlarged (presumably due to

Biotope name	Recoverability
	wave action and predation), with little recovery within 3-5 years, leading to estimated recovery times of 8-34 years (Pain & Levin, 1981) or several hundreds of years (Seed & Suchanek, 1992). *Mytilus edulis* populations were considered to have a strong ability to recover from environmental disturbance (Holt *et al.*, 1998; Seed & Suchanek, 1992). While good annual recruitment is possible, recovery may take at least 5 years but significantly longer in certain circumstances and some environmental conditions. Bennell (1981) observed that barnacle populations removed when the surface rock was scraped off in a barge accident at Amlwch, North Wales returned to pre-accident levels within 3 years. However, barnacle recruitment can be very variable because it is dependent on a suite of environmental and biological factors, such as wind direction and the presence of adults as an inducement for larvae to settle, therefore populations may take longer to recover. Studies of community recovery after the *Torrey Canyon* oil spill have suggested that sites affected by oil alone recovered rapidly, within 3 years. However, areas that were treated with dispersants resulted in death of a large number of invertebrates, especially grazing gastropods and barnacles due to smothering by the subsequent bloom of macroalgae. Recovery was variable, with heavily impacted areas taking 2->10 years to regain their prior species richness, whilst many shores recovered in 5-8 years while it was estimated that the worst affected shores would take ca 15 years (Southward & Southward, 1978; Hawkins & Southward, 1992; Raffaelli & Hawkins, 1999). Recruitment in mobile species may be rapid once the mussel matrix develops. Macroalgae such as ephemeral greens (e.g. *Ulva* spp.) with recruit rapidly and *Fucus vesiculosus* recruits readily to cleared areas of the shore and full recovery takes 1-3 years (Holt *et al.*, 1997). Overall, the rate of recovery is probably dependant on the degree and nature of the disturbance. Where the dominant species (barnacles and mussels) are reduced or removed but other members of the community remain or adults can recruit from the surrounding area, recovery will probably be rapid, within ca 5 years. Similarly, if other members of the community alone are reduced or removed (e.g. mobile epifauna or macroalgae) recovery is likely to be rapid. However, recovery from a significant impact, especially the mass mortality of grazing gastropods, is likely to result in marked affects on the community that may take between 5-10 years to recover and in some cases about 15 years.
Barnacles and *Patella* spp. on exposed or moderately exposed, or vertical sheltered, eulittoral rock (ELR.BPat)	Bennell (1981) observed that barnacle populations removed when the surface rock was scraped off in a barge accident at Amlwch, North Wales returned to pre-accident levels within 3 years. However, barnacle recruitment can be very variable because it is dependent on a suite of environmental and biological factors, such as wind direction and the presence of adults as an inducement for larvae to settle, therefore populations may take longer to recover. Recolonization of *Patella vulgata* on rocky shores is rapid as seen by the appearance of limpet spat 6 months after the *Torrey Canyon* oil spill reaching peak numbers 4-5 years after the spill (Southward & Southward, 1978). Most characterizing species have planktonic larvae and/or are mobile and so can migrate into the affected area. Therefore, it seems likely that within five years the community should be able to return to a pre-impact state so recovery is set to high.

Biotope name	Recoverability
Corallina officinalis on very exposed lower eulittoral rock (ELR.Coff)	*Corallina officinalis* probably has good recruitment and settled on artificial substrata within 1 week of their placement in the intertidal during summer in New England (Harlin & Lindbergh 1977). New fronds of *Corallina officinalis* appeared on sterilized plots within six months and 10% cover was reached with 12 months (Littler & Kauker 1984). Bamber & Irving (1993) reported that new plants grew back in scraped transects within 12 months, although the resistant crustose bases were probably not removed. Similarly, in experimental plots, up to 15% cover of *Corallina officinalis* fronds returned within 3 months after removal of fronds and all other epiflora/fauna but not the crustose bases (Littler & Kauker 1984). Although new crustose bases may recruit and develop quickly the formation of new fronds from these bases and recovery of original cover may take longer, and it is suggested that the population is likely to recover within a few years. If the holdfasts of red algae remain, they are likely to recover quickly, as if damaged by winter storms. For example, following experimental harvesting by drag raking in New Hampshire, USA, populations of *Chondrus crispus* recovered to one third of their original biomass after 6 months and totally recovered after 12 months (Mathieson & Burns, 1975). Raking is designed to remove the large fronds but leave the small upright shoots and holdfasts. The authors suggested that control levels of biomass and reproductive capacity are probably re-established after 18 months of regrowth, although time to recovery was much extended if harvesting occurred in the winter, rather than the spring or summer (Mathieson & Burns, 1975). Minchinton *et al.* (1997) documented the recovery of *Chondrus crispus* after a rocky shore in Nova Scotia, Canada, was totally denuded by an ice scouring event. Initial recolonization was dominated by diatoms and ephemeral macroalgae, followed by fucoids and then perennial red seaweeds. After 2 years, *Chondrus crispus* had re-established approximately 50% cover on the lower shore and after 5 years it was the dominant macroalgae at this height, with approximately 100% cover. Therefore, recovery by *Chondrus crispus* will be relatively rapid (approximately 18 months) in situations where intolerance to a factor is intermediate and some holdfasts remain for regeneration of fronds. In situations of high intolerance, where the entire population of *Chondrus crispus* is removed, recovery will be limited by recruitment from a remote population and would be likely to take up to 5 years. Similarly, clearance studies of concrete blocks in the shallow subtidal showed that Rhodophyceae colonized and grew in the winter months, presumably at their peak of spore availability (Kain, 1975). It is probably that most of the characterizing red algae would grow back from remaining holdfasts and recruit well in winter and where bases are removed they will probably take a few years to regain their original cover, although in this biotope their percentage cover is low. The epiphytic fauna are mainly composed of mobile species that will recruit quickly from surrounding habitats, and will therefore, recover quickly once the coralline turf has developed. Overall, where upright fronds of the red algal turf are removed, recovery will probably be very rapid, within about 12 months. If the holdfasts are removed, recovery of their original cover is likely to be prolonged but the biotope would probably be recognizable within less than 5 years.

Biotope name	Recoverability
Himanthalia elongata and red seaweeds on exposed lower eulittoral rock (ELR.Him)	For all algal species within the biotope, recovery periods would vary with season owing to the availability of spores and the proximity of fertile specimens to denuded areas, and /or the presence of vegetative material from which new plants could propagate. Furthermore, spores of red algae are not motile so algal dispersal is wholly a passive process (Fletcher & Callow, 1992) and colonization may rely on the presence of reproducing plants only a few metres away. *Himanthalia elongata* recruited to a suitable substratum (placed in the eulittoral amongst adjacent species) within one year but initial plant densities declined to three or four holdfasts owing to the lack of protection from established adults or other foliose algae which would have provided protection from desiccation, wave action and high irradiance. However, the number of holdfasts rose to 1500 buttons per block by March of the second year (Stengel *et al.*, 1999). A recovery time of more than three years was reported by MacFarlane (1952) and Mathieson & Burns (1975) following total removal of *Chondrus crispus* by scraping. However, based on experimental evidence, regrowth of *Mastocarpus stellatus* and *Chondrus crispus* is likely to be good in instances where some holdfast is left intact, with recovery to pre-removal abundance of both species occurring within one year, and often considerably less, in lightly harvested areas, but taking 1-2 years in areas more heavily cleared (but with some material remaining) (Marshall *et al.*, 1949 and other citations in Holt *et al.*, 1995). In kelp canopy removal experiments in the Isle of Man, Hawkins & Harkin (1985) observed a rapid increase in the number of *Palmaria palmata* sporelings on bare rock and the species came to dominate cleared plots within five months. Recolonization from distant populations would probably take longer, however, because dispersal distances are limited, with spores sinking and attaching close to adult plants. Recolonization of *Patella vulgata* on rocky shores is likely to be rapid as it is a cosmopolitan species with planktonic dispersal. For instance, limpet spat recruited to suitable substratum within 6 months after the Torrey Canyon oil spill, peak abundance of the species was noted within 4-5 years after the spill. Bennell (1981) observed that *Semibalanus balanoides* were removed when the surface rock was scraped off in a barge accident at Amlwch, North Wales. Barnacle populations returned to pre-accident levels within 3 years. However, barnacle recruitment can be very variable because it is dependent on a suite of environmental and biological factors (see full *MarLIN* review: reproduction), therefore barnacle populations may take longer to recover. Many other faunal species in the biotope are widespread and have planktonic life stages which would aid recovery. Mobile species such as *Nucella lapillus* (which lays eggs producing mobile young) are capable of recovering with about 2-5 years if survivors are present nearby intertidally or below low water. However, should a population need to recruit from distant locations recovery may take significantly longer as the species are relatively slow crawlers. Recoverability of the biotope has been assessed to be high if fertile populations of both faunal and floral species are in the immediate vicinity or if, in the case of algae, vegetative material remains in situ. In the absence of either of the aforementioned, partial recovery may occur within 5 years but full recovery could take longer.

Biotope name	Recoverability
Barnacles and fucoids (moderately exposed shores) (MLR.BF)	Recovery is high because recruitment of key species, with the exception of *Ascophyllum nodosum*, is fairly rapid so that the biotope will look much as before within five years. Most characterizing species have planktonic larvae and/or are mobile and so can migrate into the affected area. However, although the recruitment of many species in the barnacle-fucoid mosaics is rapid, it can take between 10 and 15 years for the natural variation in community structure of the biotope to return to normal after significant mortality of key species such as seen after the *Torrey Canyon* oil spill (Southward & Southward, 1978).
Underboulder communities (MLR.Fser.Fser.Bo)	The community associated with MLR.Fser.Fser.Bo will very greatly depending on various factors including the size of boulder, wave exposure and the presence or absence of flowing water under the boulder. In addition, it is difficult to identify a 'climax' community *per se* because the extent of community succession will vary greatly between boulders of different sizes etc. Furthermore, because there are no key functional, structural or characterizing species, any combination of the important species could, theoretically, determine the biotope community. Nevertheless, the recolonization of fauna typically associated with MLR.Fser.Fser.Bo will occur within a year or two and recoverability is expected to be high. However, the development of a mature community characteristic of seldom disturbed boulders dominated by e.g. *Halichondria panicea* and *Dendrodoa grossularia* may take longer although many boulders will never mature to this stage. In the study of recolonization of vertical rock wall in Maine (Sebens, 1986), epifaunal and algal crust species were shown to re-colonize cleared areas quickly. For example encrusting bryozoans, tubeworms, tubicolous amphipods and worms, erect hydroids and bryozoans were reported to cover cleared areas within 1-4 months in spring, summer and autumn (Sebens, 1986). Sebens (1985) reported that *Halichondria panicea* had reached previous cover within two or more years. It was slow to recolonize the cleared areas, only appearing after about a year, although it is relatively fast growing. *Balanus crenatus* is another important early colonizer of sublittoral rock surfaces (Kitching, 1937) and it heavily colonized a site that was dredged for gravel within 7 months (Kenny & Rees, 1994).
Ulva spp. on freshwater-influenced or unstable upper eulittoral rock (MLR.Ent)	The biotope is considered to have a very high recoverability following disturbance. *Ulva intestinalis* is generally considered to be an opportunistic species, with an 'r-type' strategy for survival. The r-strategists have a high growth rate and high reproductive rate. For instance, the thalli of *Ulva intestinalis*, which arise from spores and zygotes, grow within a few weeks into thalli that reproduce again, and the majority of the cell contents are converted into reproductive cells. The species is also capable of dispersal over a considerable distance. For instance, Amsler & Searles (1980) showed that 'swarmers' of a coastal population of *Ulva* (as *Enteromorpha*) reached exposed artificial substrata on a submarine plateau 35 km away. *Ulva* species are amongst the first multicellular algae to appear on substrata that have been cleared following a disturbance, e.g. following the *Torrey Canyon* oil spill in March 1967, species of the genus *Ulva* rapidly recruited to areas where oil had killed the herbivores that usually grazed on them, so that a rapid greening of the rocks (owing to a thick coating of *Ulva*) was apparent by mid-May (Smith, 1968). *Porphyra* is also able to rapidly recruit to cleared substrata, and may regenerate from its discoid shaped holdfast if it remains in situ. After the

Biotope name	Recoverability
	Torrey Canyon oil spill, its presence was noted on rocks within two months of the disturbance (see 'general biology' time to reach maturity).
Rhodothamniella floridula on sand-scoured lower eulittoral rock (MLR.Rho)	Following major loss of component species, there will be a colonization succession such as that observed following a severe oil spill. The loss of grazing species results in an initial proliferation of ephemeral green then fucoid algae, which then attracts mobile grazers, and encourages settlement of other grazers. Limpet grazing reduces the abundance of fucoids allowing barnacles to colonize the shore. Recovery rates were dependant on local variation in recruitment and mortality so that sites varied in recovery rates, for example maximum cover of fucoids occurred within 1-3 years, barnacle abundance increased in 1-7 years, limpet number were still reduced after 6-8 years and species richness was regained in 2 to >10 years. Overall, recovery took 5-8 years on many shores but was estimated to take about 15 years on the worst affected shores (Southward & Southward, 1978; Hawkins & Southward, 1992; Raffaelli & Hawkins, 1999). This biotope is characterized by mats of *Rhodothamniella floridula*. No information was found relating to colonization or recolonization rates of *Rhodothamniella floridula*, however, red algae are typically highly fecund, but their spores are non-motile (Norton, 1992) and therefore highly reliant on the hydrodynamic regime for dispersal. Kain (1975) reported that after displacement some Rhodophyceae were present after 11 weeks, and after 41 weeks, in June, Rhodophyceae species predominated. However, Stegenga (1978) noted that tetrasporangia of *Rhodothamniella floridula* (as *Rhodochorton floridulum*) germinated in 'rather low numbers'. Recoverability of the biotope has been assessed as high, although recovery of remote populations will be more protracted and dependent upon favourable currents bringing spores.
Mytilus edulis and *Fucus vesiculosus* on moderately exposed mid eulittoral rock (MLR.MytFves)	Macroalgae such as ephemeral greens (e.g. *Ulva* spp.) with recruit rapidly and *Fucus vesiculosus* recruits readily to cleared areas of the shore and full recovery takes 1-3 years (Holt *et al.*, 1997). *Mytilus edulis* is highly fecund but larval mortality is high. Larval development occurs within the plankton over ca 1 month (or more), with high dispersal potential. Recruitment within the population is possible when larvae may be entrained within enclosed coasts but it is likely that larval produced in open coast examples of the biotope are swept away from the biotope to settle elsewhere. Larval supply and settlement could potentially occur annually but settlement is sporadic with unpredictable pulses of recruitment (Lutz & Kennish, 1992; Seed & Suchanek, 1992). Once settled, *Mytilus edulis* can reproduce within its first year if growth conditions allow. On rocky shores, gaps in beds of mussels are often colonized by barnacles and fucoids, barnacles enhancing subsequent recruitment of mussels. The presence of macroalgae in disturbance gaps in *Mytilus califorianus* populations, where grazers were excluded, inhibited recovery by the mussels. In New England, U.S.A, prior barnacle cover was found to enhance recovery by *Mytilus edulis* (Seed & Suchanek, 1992). Cycles of loss and recruitment leads to a patchy distribution of mussels on rocky shores. High intertidal and less exposed sites recovered slower than low shore, more exposed sites. Several long term studies showed that gaps in mussel beds took a long time to heal, but in some cases enlarged (presumably due to wave action and predation), with little recovery within 3-5 years, leading to estimated recovery times of 8-34 years (Pain & Levin, 1981) or several hundreds of years (Seed & Suchanek, 1992). Recruitment

Biotope name	Recoverability
	in mobile species may be rapid once the mussel matrix develops. Development of the community from bare or denuded rock is likely to follow a similar succession to that occurring after an oil spill. The loss of grazing species results in an initial proliferation of ephemeral green then fucoid algae, which then attracts mobile grazers, and encourages settlement of other grazers. Limpet grazing reduces the abundance of fucoids allowing barnacles to colonize the shore. Recovery of rocky shore populations was intensively studied after the *Torrey Canyon* oil spill in March 1967. Areas affected by oil alone recovered rapidly, within 3 years. But other sites suffered substantial damage due to the spilled oil and the application of aromatic hydrocarbon based dispersants. In the latter sites, populations of fucoids were abnormal for the first 11 years, and limpet *Patella vulgata* populations were abnormal for at least 10-13 years. Recovery rates were dependant on local variation in recruitment and mortality so that sites varied in recovery rates, for example maximum cover of fucoids occurred within 1-3 years, barnacle abundance increased in 1-7 years, limpet number were still reduced after 6-8 years and species richness was regained in 2 to >10 years. Overall, recovery took 5-8 years on many shores but was estimated to take about 15 years on the worst affected shores (Southward & Southward, 1978; Hawkins & Southward, 1992; Raffaelli & Hawkins, 1999). This biotope is characterized by the presence of dense *Mytilus edulis*. *Mytilus* spp. populations were considered to have a strong ability to recover from environmental disturbance (Holt *et al.*, 1998; Seed & Suchanek, 1992). While good annual recruitment is possible, recovery of gaps in the mussel population may take up to 5 years. However, where the biotope is significantly damaged recovery of the mussel population may be delayed by 1-7 years for the initial macroalgal cover to reduce and barnacle cover to increase. Therefore, a recognizable biotope may take between 5 -10 years to recover depending on local conditions. However, it should be noted that in certain circumstances and under some environmental conditions recovery may take significantly longer.
Sabellaria alveolata reefs on sand-abraded eulittoral rock (MLR.Salv)	Although reproduction occurs each year, recruitment in *Sabellaria alveolata* is very sporadic and unpredictable which means that recovery could take several years. The presence of remaining adults will assist in larval settlement, as this is the preferred substratum (Wilson, 1929). If further recruitment does not then occur, allowing new growth, the reef will disintegrate. There is no real 'mature stage' as such, rather a cycle of growth and decay. Recovery from factors to which MLR.Salv has a high intolerance is likely to be moderate.
Ascophyllum nodosum on very sheltered mid eulittoral rock (SLR.Asc)	Where *Ascophyllum nodosum* is lost, recovery would be slow due to the slow growth rate and poor recruitment of this dominant species and so a rank of moderate recovery is reported. The lack of recovery of *Ascophyllum nodosum* from harvesting is well documented. For example, in their work on fucoid recolonization of cleared areas at Port Erin, Knight and Parke (1950) observed that even eight years after the original clearance there was still no sign of the establishment of an *Ascophyllum nodosum* population. The species is extremely fertile every year and Printz (1959) suggests it must be assumed that some special combination of climatic or environmental conditions is needed for an effective recolonization. However, most associated species are likely to recover fairly rapidly (within two to five years) due to recruitment from planktonic larvae or through immigration. Thus, if *Ascophyllum nodosum* remains recovery of the biotope will be much more rapid and a rank of high is reported.

Biotope name	Recoverability
Fucus ceranoides on reduced salinity eulittoral rock (SLR.Fcer)	Recolonization of cleared *Fucus*-dominated areas may take between one to three years in British waters, with the exception of *Ascophyllum nodosum*, and is especially rapid in areas cleared of grazers (Hartnoll & Hawkins, 1985; Hawkins & Hartnoll, 1985), though it may take longer before normal cycles of stability are reached. *Ulva* spp. is an ephemeral seaweed that is amongst the first to colonize newly available substrate, usually within weeks, depending upon availability of spores. Prosobranchs are represented by *Littorina littorea*, which is widespread and often common or abundant. Adults are slow crawlers so active immigration of snails is unlikely. Recolonization may occur through rafting of adults on floating wood or weed. The larvae form the main mode of dispersal. *Littorina littorea* is an iteroparous breeder with high fecundity that lives for several (at least 4) years. Breeding can occur throughout the year and the planktonic larval stage is long (up to 6 weeks). Bennell (1981) observed that following accidental removal of barnacles at Amlwch, North Wales, the barnacle populations returned to pre-accident levels within 3 years. However, barnacle recruitment can be very variable because it is dependent on a suite of environmental and biological factors, such as wind direction (see reproduction, in *MarLIN* review of *Semibalanus balanoides*). *Elminius modestus* produces several broods each year. With an initial growth rate over 40 days of ca 6 mm, *Elminius modestus* reaches maturity in its first season (Eno, 1997). Recolonization, recruitment and recovery rates for the majority of species particularly characteristic of the biotope should therefore be high (within five years).
Barnacles and *Littorina littorea* on unstable eulittoral mixed substrata (SLR.BLlit)	Recovery of the biotope should be high as most species are abundant and widely distributed, are iteroparous and have planktonic larvae. For example, Bennell (1981) observed that barnacle populations removed when the surface rock was scraped off in a barge accident at Amlwch, North Wales returned to pre-accident levels within 3 years. However, barnacle recruitment can be very variable because it is dependent on a suite of environmental and biological factors, such as wind direction and the presence of adults as an inducement for larvae to settle, therefore populations may take longer to recover. *Littorina littorea* is widespread and often common or abundant. *Littorina littorea* is an iteroparous breeder with high fecundity that lives for several (at least 4) years. Breeding can occur throughout the year. The larvae form the main mode of dispersal with a long planktonic stage (up to 6 weeks) although larvae do tend to remain in waters close to the shore. Most of the other species in the biotope have planktonic larvae and so can recolonize the affected area. Therefore, it seems likely that within five years the community should be able to return to a pre-impact state so recovery is set to high. However, in cases where grazing prosobranchs are lost there is likely to be growth of first green and then brown algae which may then come to dominate the shore until removed by scour or old age. In such cases the re-establishment of SLR.Bllit may take longer than five years and a rank of moderate has been given.
Fucus vesiculosus on mid eulittoral mixed substrata (SLR.FvesX)	*Fucus vesiculosus* recruits readily to cleared areas of the shore and full recovery takes 1-3 years in British waters (Hartnoll & Hawkins, 1985; Hawkins & Hartnoll, 1985). Faunal species of the biotope are widespread and fecund with a planktonic larval stage so dispersion can occur over some distance. Recoverability of the SLR.Asc.X biotope will differ from that of the SLR.FvesX and SLR.FserX biotope. Where the whole plant is removed,

Biotope name	Recoverability
	recovery is slow due to the slow growth rate and poor recruitment of *Ascophyllum nodosum*. In their work on fucoid recolonization of cleared areas at Port Erin, Knight and Parke (1950) observed that even eight years after the original clearance there was still no sign of the establishment of an *Ascophyllum nodosum* population. Even in an area where many plants remained after harvesting no repopulation was seen for several years (Printz, 1959). The species is extremely fertile every year and Printz (1959) suggested that some special combination of climatic or environmental conditions is needed for an effective recolonization. Recovery of the population to original abundance and biomass is likely to take a very long time and has been assessed to be typically low.
Green seaweeds (*Ulva* spp. and *Cladophora* spp.) in upper shore rockpools (LR.G)	It is likely that *Ulva* and *Cladophora* species will have a considerable capacity for recovery. Both species are widespread around the British Isles and Ireland, and may be found in reproductive condition all year round. Numerous motile swarmers (gametes and spores) are released and in the water column they can be dispersed over considerable distances. In addition to recruitment by swarmers, new growth of *Cladophora rupestris* may arise from the resistant multicellular branching rhizoids (van den Hoek, 1982) that may remain in situ. Recoverability has therefore been assessed to be very high. For instance, after the *Torrey Canyon* tanker oil spill in mid March 1967, recolonization by sporelings of *Ulva* and *Cladophora* species had occurred by the end of April (Smith, 1968). Recovery of the copepod *Tigriopus fulvus* would be expected to be rapid (presuming a residual or localized population remained from which to recruit) as the species is in reproductive condition all year round and reaches sexual maturity within two months. It also can produce more than one brood from one fertilization. These aforementioned species are characteristic of the biotope, which would be recognized upon their probably rapid re-establishment. Other components of the community, such as the littorinids and other grazers, are of lower abundance owing to physical conditions and are not considered to be characterizing species. However, in their total absence the biotope would be considered to be impoverished. Owing to recruitment of live young in a localized area without a dispersive larval stage recovery of such species may take longer.
Corallina officinalis and coralline crusts in shallow eulittoral rockpools (LR.Cor)	*Corallina officinalis* probably has good recruitment and settled on artificial substances within 1 week of their placement in the intertidal in New England summer (Harlin & Lindbergh, 1977). New fronds of *Corallina officinalis* appeared on sterilized plots within six months and 10% cover was reached with 12 months (Littler & Kauker, 1984). Bamber & Irving (1993) reported that new plants grew back in scraped transects within 12 months, although the resistant crustose bases were probably not removed. Similarly, in experimental plots, up to 15% cover of *Corallina officinalis* fronds returned within 3 months after removal of fronds and all other epiflora/fauna but not the crustose bases (Littler & Kauker, 1984). Although new crustose bases may recruit and develop quickly the formation of new fronds from these bases and recovery of original cover may take longer, and it is suggested that a population is likely to recover within a few years. Chamberlain (1996) observed that although *Lithophyllum incrustans* was quickly affected by oil during the *Sea Empress* spill, recovery occurred within about a year. The oil was found to have destroyed about one third of the thallus thickness but regeneration occurred from thallus filaments below the damaged area. A recoverability of high is therefore suggested. If colonies were completely destroyed new growth would be slow and, because of low growth rates, recoverability would be low. If death occurred,

Biotope name	Recoverability
	recoverability will be slow. Spores will settle and new colonies will arise rapidly on bare substratum but growth rate is slow (2-7 mm per annum - see Irvine & Chamberlain 1994). Recolonization of *Patella vulgata* on rocky shores is rapid as seen by the appearance of limpet spat 6 months after the *Torrey Canyon* oil spill reaching peak numbers 4-5 years after the spill (Southward & Southward, 1978). The epifauna are mainly composed of mobile species that will recruit quickly from surrounding habitats, and will therefore, recover quickly once the coralline turf has developed.
Fucoids and kelps in deep eulittoral rockpools (LR.FK)	Kain (1975) examined recolonization of cleared concrete blocks in a subtidal kelp forest. Red algae colonized blocks within 26 weeks in the shallow subtidal (0.8m) and 33 weeks at 4.4m. After about 2.5 years, *Laminaria hyperborea* standing crop, together with an understorey of red algae, was similar to that of virgin forest. Red algae were present throughout the succession increasing from 0.04 to 1.5 percent of the biomass within the first 4 years. Colonizing species varied with time of year, for example blocks cleared in August 1969 were colonized by primarily *Laminaria saccharina* and subsequent colonization by *Laminaria hyperborea* and other laminarians was faster than blocks colonized by *Saccorhiza polyschides*; within 1 year the block was occupied by laminarians and red algae only. Succession was similar at 4.4m, and *Laminaria hyperborea* dominated within about 3 years. Blocks cleared in August 1969 at 4.4m were not colonized by *Saccorhiza polyschides* but were dominated by red algae after 41 weeks, e.g. *Cryptopleura ramosa*. Kain (1975) cleared one group of blocks at two monthly intervals and noted that brown algae were dominant colonists in spring, green algae (solely *%*) in summer and red algae were most important in autumn and winter. Overall, red algae are likely to be able to recolonize and recover abundance with a year in some instances and probably within 5 years. Similarly, laminarians could potentially colonize low shore rockpools within 3-4 years, depending on grazing and competition for space. Red algae produce non motile spores, dependant on the hydrography and most recruitment is likely to occur within about 10 m of the parent plants (Norton, 1992). Therefore, within a rock pool or a pool surrounded by macroalgae, recruitment is likely to be good. However, recruitment from remote populations is likely to be more protracted and sporadic. Recovery of a population of *Chondrus crispus* following a perturbation is likely to be largely dependent on whether holdfasts remain, from which new thalli can regenerate (Holt *et al.*, 1995). Following experimental harvesting by drag raking in New Hampshire, USA, populations recovered to 1/3 of their original biomass after 6 months and totally recovered after 12 months (Mathieson & Burns, 1975). Raking is designed to remove the large fronds but leave the small upright shoots and holdfasts. The authors suggested that control levels of biomass and reproductive capacity are probably re-established after 18 months of regrowth. It was noted however, that time to recovery was much extended if harvesting occurred in the winter, rather than the spring or summer (Mathieson & Burns, 1975). Minchinton *et al.* (1997) documented the recovery of *Chondrus crispus* after a rocky shore in Nova Scotia, Canada, was totally denuded by an ice scouring event. Initial recolonization was dominated by diatoms and ephemeral macroalgae, followed by fucoids and then perennial red seaweeds. After 2 years, *Chondrus crispus* had re-established approximately 50% cover on the lower shore and after 5 years it was the dominant macroalga at this height, with approximately 100% cover. The authors pointed out that although

Biotope name	Recoverability
	Chondrus crispus was a poor colonizer, it was the best competitor. Fucoids (e.g. *Fucus serratus* and *Fucus vesiculosus*) recruit readily to cleared areas, especially in the absence of grazers (Holt *et al.*, 1997). However, fucoid propagules tend to settle near to the parent plants, due to turbulent deposition by water flow. Within monospecific stands recruitment of conspecifics is most likely, and community recovery is likely to be rapid. For example, after the *Torrey Canyon* oil spill, fucoids attained maximum cover within 1-3 years (Southward & Southward, 1978; Hawkins & Southward, 1992; Raffaelli & Hawkins, 1999). However, in cleared areas, recruitment is likely to be rapid but recovery of the original community structure is likely to take some years (Holt *et al.*, 1997). For example, after the *Torrey Canyon* oil spill, although maximum cover of fucoids occurred within 1-3 years, the abundance of barnacles increased in 1-7 years, limpet number were still reduced after 6-8 years and species richness was regained in 2 to >10 years (Southward & Southward, 1978; Hawkins & Southward, 1992; Raffaelli & Hawkins, 1999). Sousa *et al.* (1981) reported that experimental removal of sea urchins significantly increased recruitment in long-lived brown algae. In experimental plots cleared of algae and sea urchins in December, *Halidrys dioica* colonized the plots, in small numbers, within 3-4 months. Plots cleared in August received few, if any recruits, suggesting that recolonization was dependant on zygote availability and therefore the season. Wernberg *et al*. (2001) suggested that the lack of long range dispersal success in *Halidrys siliquosa* was responsible for its regional distribution in the north east Atlantic *Corallina officinalis* probably has good recruitment and settled on artificial substrata within 1 week of their placement in the intertidal during summer in New England (Harlin & Lindbergh, 1977). New fronds of *Corallina officinalis* appeared on sterilized plots within six months and 10% cover was reached with 12 months (Littler & Kauker 1984). Bamber & Irving (1993) reported that new plants grew back in scraped transects within 12 months, although the resistant crustose bases were probably not removed. Similarly, in experimental plots, up to 15% cover of *Corallina officinalis* fronds returned within 3 months after removal of fronds and all other epiflora/fauna but not the crustose bases (Littler & Kauker, 1984). Although new crustose bases may recruit and develop quickly the formation of new fronds from these bases and recovery of original cover may take longer, and it is suggested that the population is likely to recover within a few years. Gastropods and other mobile grazers (e.g. amphipods, isopods) are likely to be attracted by developing microalgae and macroalgae and could return quickly by either migration or larval recruitment. Epifaunal species vary in their recruitment rates. Sebens (1985, 1986) reported that rapid colonizers such as encrusting corallines, encrusting bryozoans, amphipods and tubeworms recolonized cleared rock surfaces within 1-4 months. Ascidians such as *Aplidium* spp. achieved significant cover in less than a year, and, together with *Halichondria panicea*, reached pre-clearance levels of cover after 2 years. Anemones colonized within 4 years (Sebens, 1986) and would probably take longer to reach pre-clearance levels. The anemone *Urticina felina* has poor powers of recoverability due to poor dispersal (Sole-Cava *et al.*, 1994 for the similar *Tealia crassicornis*) and slow growth (Chia & Spaulding, 1972), though populations should recover within 5 years. Overall, members of the rockpool community could potentially recolonize with a year and a recognizable biotope return within 5 years. However,

Biotope name	Recoverability
	rockpool recruitment is reported to be sporadic and variable (Metaxas & Scheibling, 1993). While a recognizable biotope will return the exact community may differ from that present prior to perturbation. In addition, although the biotope is likely to be recognizable within less than 5 years, if the community was completely destroyed by perturbation, it may take longer for a typically diverse community to become established, especially the biotopes supported anemones and the rarer red algal species.
Seaweeds in sediment (sand or gravel)-floored eulittoral rockpools (LR.SwSed)	Red algae produce non motile spores, dependant on the hydrography and most recruitment is likely to occur within about 10 m of the parent plants (Norton, 1992). Therefore, within a rock pool or a pool surrounded by macroalgae, recruitment is likely to be good. However, recruitment from remote populations is likely to be more protracted and sporadic. The life history characteristics of *Ahnfeltia plicata* suggest that the species is likely to recover within 5 years if local populations exist (see *MarLIN* review). Recovery of a population of *Chondrus crispus* following a perturbation is likely to be largely dependent on whether holdfasts remain, from which new thalli can regenerate (Holt *et al.*, 1995). Following experimental harvesting by drag raking in New Hampshire, USA, populations recovered to 1/3 of their original biomass after 6 months and totally recovered after 12 months (Mathieson & Burns, 1975). Raking is designed to remove the large fronds but leave the small upright shoots and holdfasts. The authors suggested that control levels of biomass and reproductive capacity are probably re-established after 18 months of regrowth. It was noted however, that time to recovery was much extended if harvesting occurred in the winter, rather than the spring or summer (Mathieson & Burns, 1975). Minchinton *et al.* (1997) documented the recovery of *Chondrus crispus* after a rocky shore in Nova Scotia, Canada, was totally denuded by an ice scouring event. Initial recolonization was dominated by diatoms and ephemeral macroalgae, followed by fucoids and then perennial red seaweeds. After 2 years, *Chondrus crispus* had re-established approximately 50% cover on the lower shore and after 5 years it was the dominant macroalga at this height, with approximately 100% cover. The authors pointed out that although *Chondrus crispus* was a poor colonizer, it was the best competitor. Kain (1975) examined recolonization of cleared concrete blocks in a subtidal kelp forest. Red algae colonized blocks within 26 weeks in the shallow subtidal (0.8m) and 33 weeks at 4.4m. After about 2.5 years, *Laminaria hyperborea* standing crop, together with an understorey of red algae, was similar to that of virgin forest. Red algae were present throughout the succession increasing from 0.04 to 1.5 percent of the biomass within the first 4 years. Colonizing species varied with time of year, for example blocks cleared in August 1969 were colonized by primarily *Laminaria saccharina* and subsequent colonization by *Laminaria hyperborea* and other laminarians was faster than blocks colonized by *Saccorhiza polyschides*; within 1 year the block was occupied by laminarians and red algae only. Succession was similar at 4.4m, and *Laminaria hyperborea* dominated within about 3 years. Blocks cleared in August 1969 at 4.4m were not colonized by *Saccorhiza polyschides* but were dominated by red algae after 41 weeks, e.g. *Cryptopleura ramosa*. Kain (1975) cleared one group of blocks at two monthly intervals and noted that brown algae were dominant colonists in spring, green algae (solely %) in summer and red algae were most important in autumn and winter. Overall, red algae are likely to be able to recolonize and recover abundance with a year in some instances and probably within 5 years. Similarly,

Biotope name	Recoverability
	laminarians could potentially colonize low shore rockpools within 3-4 years, depending on grazing and competition for space. Fucoids (e.g. *Fucus serratus*) are highly fecund, reproduce throughout the years, are widespread and could potentially recovery quickly. For example, after the *Torrey Canyon*, oil spill fucoids attained maximum cover within 1-3 years (Southward & Southward, 1978; Hawkins & Southward, 1992; Raffaelli & Hawkins, 1999). *Furcellaria lumbricalis* is an exception. Although highly fecund (Austin, 1960a), the species grows very slowly compared to other red algae (Bird *et al*., 1991) and takes a long time to reach maturity, typically 5 years (Austin, 1960b). Christensen (1971; cited in Bird *et al.*, 1991) noted that following harvesting of *Furcellaria lumbricalis* forma *aegagropila* in the Baltic Sea, harvestable biomass had not been regained 5 years after the suspension of harvesting. In view of its slow growth, time to maturity and limited dispersal, recovery of *Furcellaria lumbricalis* is likely to take between 5 and 10 years to recover in situations where intolerance to a factor is high. Where a portion of the population remains for vegetative regrowth, recovery is likely to occur within 5 years. Gastropods and other mobile grazers (e.g. amphipods, isopods) are likely to be attracted by developing microalgae and macroalgae and could return quickly by either migration or larval recruitment. Epifaunal species vary in their recruitment rates. Sebens (1985, 1986) reported that rapid colonizers such as encrusting corallines, encrusting bryozoans, amphipods and tubeworms recolonized cleared rock surfaces within 1-4 months. Ascidians such as *Aplidium* spp. achieved significant cover in less than a year, and, together with *Halichondria panicea*, reached pre-clearance levels of cover after 2 years. Anemones colonized within 4 years (Sebens, 1986) and would probably take longer to reach pre-clearance levels. The anemone *Urticina felina* has poor powers of recoverability due to poor dispersal (Sole-Cava *et al.*, 1994 for the similar *Tealia crassicornis*) and slow growth (Chia & Spaulding, 1972), though populations should recover within 5 years. Overall, members of the rockpool community could potentially recolonize with a year and a recognizable biotope return within 5 years. However, rockpool recruitment is reported to be sporadic and variable (Metaxas & Scheibling, 1993). Therefore, while a recognizable biotope will return the exact community may differ from that present prior to perturbation.
Hydroids, ephemeral seaweeds and Littorina littorea in shallow eulittoral mixed substrata pools (LR.H)	Hydroids have the ability to produce dormant resting stages that are far more resistant to environmental change than the colony itself. Therefore, although colonies may be removed or destroyed, the resting stages may survive attached to the substratum. The resting stages provide a mechanism for rapid recovery. The medusoid and planula larval stages of *Obelia longissima* potentially result in significant powers of dispersal. In addition, few species of hydroids have specific substrata requirements, many are generalists, and *Obelia longissima* has been reported from a variety of hard substrata, together with sandy habitats (Cornelius, 1992; Cornelius, 1995b). Hydroids are also capable of asexual reproduction and many species produce dormant, resting stages that are very resistant of environmental perturbation (Gili & Hughes, 1995). Rapid growth, budding and the formation of stolons allows hydroids to colonize space rapidly. Cornelius (1992) stated that *Obelia longissima* could form large colonies within a matter of weeks. Hydroids are often the first organisms to colonize available space in settlement experiments (Gili &

Biotope name	Recoverability
	Hughes, 1995). For example, hydroids were reported to colonize an experimental artificial reef within less than 6 months becoming abundant in the following year (Jensen *et al.*, 1994). In similar studies, *Obelia* species recruited to the bases of reef slabs within 3 months and the slab surfaces within 6 months of the slabs being placed in the marine environment in summer (Hatcher, 1998). In the St Lawrence Estuary, Canada, settlement plates immersed in June were colonized by *Obelia longissima* within a few months, and *Obelia longissima* was a dominant member of the epifauna until the following July (Brault & Bourget, 1985). Overall, *Obelia longissima* is likely to recover from damage very quickly. Even where the colonies are destroyed and/or removed, remaining resting stages or colony fragments, together with rapid growth and potentially good recruitment should result in rapid recovery. *Ulva intestinalis* is generally considered to be an opportunistic species, with an 'r-type' strategy for survival. The *r*-strategists have a high growth rate and high reproductive rate. The species is also capable of dispersal over a considerable distance. *Ulva intestinalis* is amongst the first multicellular algae to appear on substrata that have been cleared following a disturbance. Following the *Torrey Canyon* oil spill in March 1967, for instance, species of the genus *Ulva* rapidly recruited to areas where oil had killed the herbivores that usually grazed on them, so that a rapid greening of the rocks (owing to a thick coating of *Ulva*) was apparent by mid-May (Smith, 1968). The rapid recruitment of *Ulva* to areas cleared of herbivorous grazers was also demonstrated by Kitching & Thain (1983). Following the removal of the urchin *Paracentrotus lividus* from areas of Lough Hyne, Ireland, *Ulva* grew over the cleared area and reached a coverage of 100% within one year. Therefore, evidence suggests that *Ulva intestinalis* is likely to have a considerable ability for recovery within a year. In the common periwinkle, the larvae form the main mode of dispersal. *Littorina littorea* is an iteroparous breeder with high fecundity that lives for several years. Breeding can occur throughout the year and the planktonic larval stage is long (up to 6 weeks) although larvae do tend to remain in waters close to the shore. Therefore recruitment and subsequent recovery rates should be high. Although adult immigration is usually an unlikely means of recovery, given their slow crawling, it may be possible in LR.H due to the likelihood of similar rockpools and *Littorina littorea* population sin close proximity. Seed & Suchanek (1992) reviewed studies of recovery of 'gaps' (naturally or artificially induced) in mussel beds in *Mytilus* species. On rocky shores, gaps are often colonized by barnacles and fucoids, barnacles enhancing subsequent recruitment of mussels. Cycles of loss and recruitment leads to a patchy distribution of mussels on rocky shores. High intertidal and less exposed sites recovered slower than low shore, more exposed sites. Overall, Mytilus spp. populations were considered to have a strong ability to recover from environmental disturbance (Seed & Suchanek, 1992; Holt *et al.*, 1998). Larval supply and settlement could potentially occur annually but settlement is sporadic with unpredictable pulses of recruitment (Lutz & Kennish, 1992; Seed & Suchanek, 1992). Therefore, while good annual recruitment is possible, recovery may take at least 5 years, although in certain circumstances and under some environmental conditions recovery may take significantly longer. However *Mytilus*, although potentially of importance for binding sediment within LR.H, is not an important characteristic species. For the three

Biotope name	Recoverability
	important characterizing species (*Obelia longissima*, *Ulva intestinalis* and *Littorina littorea*), recovery is likely to be very high.
Overhangs and caves (LR.Ov)	Settlement panels, which attract similar communities to overhang habitats, may be fully colonized within about 18 months of being placed into the environment (extrapolated from Sutherland & Karlson 1977; Todd 1994). Recovery of 'mature' communities under overhangs is likely to occur within two years and there will be dynamic stability where the same species are always present but individuals and colonies die and recruit with time.
Rhodothamniella floridula in upper littoral fringe soft rock caves (LR.RhoCv)	Following a major loss of the main characterizing species there will be a colonization succession. No information was found relating to colonization or recolonization rates of *Rhodothamniella floridula*, however, sand-binding algal species, such as *Rhodothamniella floridula*are able to colonize soft or crumbly rock more successfully than fucoids (Lewis, 1964). Red algae are typically highly fecund, but their spores are non-motile (Norton, 1992) and therefore highly reliant on the hydrodynamic regime for dispersal. Kain (1975) reported that after displacement some Rhodophyceae were present after 11 weeks, and after 41 weeks, in June, Rhodophyceae species predominated. However, Stegenga (1978) noted that tetrasporangia of *Rhodothamniella floridula* (as *Rhodochorton floridulum*) germinated in 'rather low numbers'. Recoverability of the biotope has been assessed as high, although recovery of remote populations will be more protracted and dependent upon favourable currents bringing spores.
Faunal crusts on wave-surged littoral cave walls (LR.FLR.CVOV.FaCr)	The community associated with this biotope is dominated by ephemeral and fast growing species that can colonize rapidly space created by wave energy. For this reason, the progression of recoverability will depend on what time of year disturbances occur. However, as mentioned previously, the LR.FLR.CVOV.FaCr community will vary both spatially and temporally and it is therefore difficulty to identify a ‘climax’ community as such. Furthermore, because there are no key functional, structural or characterizing species, any combination of faunal crust species could, theoretically, determine the biotope community. Nevertheless, the majority of the flora and fauna normally associated with LR.FLR.CVOV.FaCr will recolonize the areas within a year or two and recoverability is expected to be high. In Sebens' study of recolonization of vertical rock wall in Maine (Sebens, 1986), epifaunal and algal crust species were shown to re-colonize cleared areas quickly. For example encrusting bryozoans, tubeworms, tubicolous amphipods and worms, erect hydroids and bryozoans were reported to cover cleared areas within 1-4 months in spring, summer and autumn (Sebens, 1986). *Pomatoceros triqueter* is fairly widespread, reaches sexual maturity within 4 months (Hayward & Ryland, 1995b; Dons, 1927) and longevity has been recorded to be between 1.5 and 4 years (Hayward & Ryland, 1995b; Castric-Fey, 1983; Dons, 1927). Larvae are pelagic for about 2-3 weeks in the summer and about 2 months in the winter (Hayward & Ryland, 1995b), enabling them to disperse widely. Recovery is therefore likely to be high. *Actinia equina* is also likely to recover fairly rapidly from surrounding areas. The remaining species associated with this biotope may take longer to recover although it should still be within about three years. Bennell (1981) observed that, after barnacles were scraped off the surface rock in a barge accident at Amlwch in North Wales, barnacle populations returned to pre-accident levels within 3 years. However, barnacle recruitment can be very variable because it is dependent on a suite of environmental and biological

Biotope name	Recoverability
	factors. Jenkins *et al.* (2000) reported variation in settlement and recruitment of *Semibalanus balanoides* at all spatial scales studied (10s, 1000s of metres and 100s of km) in Sweden, the Isle of Man, southwest Ireland and southwest England and between 2 years, 1997 and 1998. Substantial variation in settlement and recruitment occurred between sites and variation in settlement explained 29 -99% of variation in recruitment across all sites, although not all variation in recruitment was explained by settlement at all sites. Sebens (1985) reported that *Halichondria panicea* had reached previous cover within two or more years. It was slow to recolonize the cleared areas, only appearing after about a year, although it is relatively fast growing. *Sabellaria alveolata* spawning occurs each July but actual recruitment levels vary considerably from year to year so recovery could take several years. The presence of some remaining adults will assist in larval settlement as this is the preferred substratum (Wilson, 1929). However, this species is only dominant south Wales and not the majority of records of this biotope. Encrusting coralline algae (e.g. *Lithothamnion* and *Phytomatolithon* took 1-2 years to recolonize cleared areas (Sebens, 1985; 1986) and with their slow growth rates probably take many years to recover their original abundance. Recoverability of *Lithophyllum incrustans* will be slow because although spores will settle and new colonies will arise rapidly on bare substratum, growth rate is slow (2-7 mm per annum - see Irvine & Chamberlain 1994).
Alaria esculenta on exposed sublittoral fringe bedrock (EIR.Ala)	*Alaria esculenta* can recolonize within a year and grows rapidly. Although *Corallina officinalis* can colonize new substrata within a week it grows slowly. Hawkins & Harkin (1985) noted that, in one canopy clearance experiment, *Alaria esculenta* attained 80 percent cover within 8 months, although dormant gametophytes of *Alaria esculenta* may have been present. Therefore it is likely that the *Alaria esculenta* canopy would return within a year, however the encrusting and articulated coralline turf and associated community would take less than 5 years to recover.
Laminaria hyperborea forest with a faunal cushion (sponges and polyclinids) and foliose red seaweeds on very exposed upper infralittoral rock (EIR.Lhyp.Fa)	Experimental clearance experiments (Kain 1979) in the Isle of Man showed that *Laminaria hyperborea* returned to near control levels of biomass within 3 years at 0.8m but that recovery was slower at 4.4m. However, Kain (1979) noted that grazing would slow recovery since, even though they did not prevent spore settlement, few sporophytes survived after 1 year in the presence of *Echinus esculentus*. These experiments did not remove the gametophyte 'seed' bank. Research on harvested populations of *Laminaria hyperborea* in Norway suggests that kelp forest biomass returned to pre-harvesting levels after 1-2 years, but that the plants were mainly small (1m) and that the age structure of the population was shifted towards younger plants. Sivertsen (1991 cited in *Birkett et al.* 1998) showed that kelp populations stabilize after about 4-5 year post-harvesting. However, re-growth was due primarily to growth of viable juveniles after harvesting. Other species characteristic of the biotope will also recover quickly. For instance, Kain (1975) examined recolonization of cleared concrete blocks in a subtidal kelp forest. Red algae colonized blocks within 26 weeks in the shallow subtidal (0.8m) and 33 weeks at 4.4m. *Delesseria sanguinea* was noted within 41 weeks (8 months) at 4.4m in one group of blocks and within 56-59 days after block clearance in another group of blocks. Sponge species, *Alcyonium digitatum* and ascidians are all known to colonize bare

Biotope name	Recoverability
	surfaces rapidly. However, advice in Norway suggests that kelp forest should be left for 7-10 years after harvesting for the kelp biomass and non-kelp species to recover (Birkett *et al.* 1998). Thus, full recovery after significant damage to the biotope is likely to rake in excess of five years and recoverability ranked as 'moderate'.
Laminaria hyperborea with dense foliose red seaweeds on exposed infralittoral rock (EIR.LhypR)	Wave exposed kelp forests with dense foliose seaweeds are likely to be more intolerant of incremental grazing presence and to reduction in wave exposure.
Foliose red seaweeds on exposed or moderately exposed lower infralittoral rock (EIR.FoR)	Red algal species particularly characteristic of the biotope would be expected to recover quickly. For instance, Kain (1975) examined recolonization of cleared concrete blocks in a subtidal kelp forest. Red algae colonized blocks within 26 weeks in the shallow subtidal (0.8m) and 33 weeks at 4.4m. *Delesseria sanguinea* was noted within 41 weeks (8 months) at 4.4m in one group of blocks and within 56-59 days after block clearance in another group of blocks. This suggests that *Delesseria sanguinea* can recolonize areas, but recolonization would be directly dependent on occurrence of reproductive season and spore availability. *Corallina officinalis* settled on artificial substances within 1 week of placement in the intertidal in New England, during summer (Harlin & Lindbergh 1977). New fronds of *Corallina officinalis* appeared on sterilized plots within six months and 10 %cover was reached with 12 months (Littler & Kauker 1984). Rhodophyceae have non-flagellate and non-motile spores that stick on contact with the substratum. Norton (1992) noted that algal spore dispersal is probably determined by currents and turbulent deposition. However, red algae produce large numbers of spores that may settle close to the adult especially where currents are reduced by an algal turf or in kelp forests. It is likely that red algae could recolonize an area from adjacent populations within a short period of time in ideal conditions but that recolonization from distant populations would probably take longer. Sponge species, *Alcyonium digitatum* and ascidians are all known to colonize bare surfaces rapidly. A short larval life and large numbers of larvae produced probably results in good local but poor long-range dispersal for bryozoans. Species of *Bugula* are opportunistic, capable of colonizing most hard substrata, and will probably colonize quickly in the vicinity of reproductive colonies, especially in the summer months in temperate waters. Once established, population abundance will probably also increase rapidly. Where the erect parts of colonies have been removed, regrowth from stolons may occur, resulting in rapid recovery. Therefore, bryozoan populations reduced in extent or abundance will probably recover within between 6 to 12 months in most cases due to local recruitment. Recoverability is likely to be slow in populations of *Urticina felina* where nearby individuals do not exist. The large size, slow growth rate and evidence from aquarium populations suggest that *Urticina felina* is long lived. Although it probably breeds each year there is no information regarding fecundity. Breeding probably does not occur until the anemone is at least 1.5 years old. Dispersal ability is considered to be poor in the similar *Urticina eques* (Solé-Cava *et al.*, 1994). The larva is most likely benthic and, although unlikely to settle for many days after release (based on work on the similar *Tealia crassicornis* for north-west USA), is unlikely to travel far. However, assuming that there are populations surviving nearby

Biotope name	Recoverability
	recruitment is likely to occur over the short distances involved but how rapidly is uncertain. The mollusc *Lacuna vincta* has an annual life cycle, is highly fecund and its long planktonic larval stage means that successful recruitment from other populations is likely. Recovery of important characterizing red algal species is likely within 1-2 years and qualitative recovery of faunal species is probable within five years assuming that populations that may provide recruits are in the vicinity. Some long-lived slow growing species that recruit infrequently, such as the sponge *Axinella dissimilis* (Hiscock, 1994) may not return for many decades if lost. However, because the biotope would be recognized within a very few years, recoverability has been assessed to be high.
Sponge crusts and anemones on wave-surged vertical infralittoral rock (EIR.SCAn)	Many of the species in this and similar biotopes are fast colonizing and almost all of the sessile species have planktonic larvae or propagules. For instance, for *Metridium senile*, Sebens (1985) suggests that the larva is lecithotrophic but has a 'pre-metamorphosis' period of months, a dispersal potential of >10,000m and a colonization rate of 5-10 years. Growth of *Metridium senile* is rapid. Bucklin (1985) working in Britain found that for *Metridium senile* f. *dianthus* fragments and for *Metridium senile* f. *pallidum* newly settled individuals, a growth rate of up to 0.6 mm and 0.8 mm in pedal diameter per day occurred respectively. Another dominant species, *Alcyonium digitatum*, spawns during December and January. Gametes are released into the water and fertilization occurs externally. The embryos are neutrally buoyant and float freely for 7 days. The embryos give rise to actively swimming lecithotrophic planulae, which may have an extended pelagic life (See below) before they eventually settle (usually within one or two further days) and metamorphose to polyps (Hartnoll, 1975; 1998). Some species might be slower to colonize and grow but it is expected that close to a full complement of species will have re-settled within five years. Sebens (1985) manipulated similar communities including by scraping rocks clear of existing fauna, and concluded that scraped areas returned to an approximation of the previous state within two years.
Laminaria digitata on moderately exposed sublittoral fringe rock (MIR.Ldig.Ldig)	Kain (1975) examined the recolonization of cleared concrete blocks by kelp plants and other algae and found that *Laminaria digitata* plants were re-established within 2 years and that red algae returned with a year. Many other characterizing species have planktonic larvae and/or are mobile and so can migrate into the affected area. However, although these species colonize the biotope quite rapidly maturity of the overall community is likely to be longer. For example, encrusting coralline algae such as *Lithophyllum incrustans* are slow growing (2-7 mm per annum - see Irvine & Chamberlain 1994) and recruitment of other species to the kelp bed may take longer. In dredged kelp beds in Norway for example, although the rock between *Laminaria hyperborea* plants was uniformly covered with coralline algae after 3 years, the more diverse community of cnidarians, bryozoans and sponges associated with coralline algae seen on undredged plots was absent (Birkett *et al*., 1998b). Within five years, however, the biotope is likely to have reached maturity and so recovery is assessed as high.
Laminaria digitata and piddocks on sublittoral fringe soft rock (MIR.Ldig.Pid)	Recovery of the main characterizing species, *Laminaria digitata,* is rapid with cleared rocks fully recolonized within two years (Kain, 1979). Most other characterizing species have a planktonic larva and/or are mobile and so can migrate into the affected area. Colonization of most species of fauna inhabiting kelp holdfast fauna in Norway were found as early as one year after kelp trawling (Christie *et al*., 1998) and on rocks the more diverse

Biotope name	Recoverability
	community of coralline algae joined by species of cnidarians, bryozoans and sponges seen on undredged plots was absent three years after kelp trawling (Birkett *et al*., 1998b). However, although full species richness and abundance may be reduced the appearance of the biotope will be much as before substratum loss and so recovery is high.
Laminaria saccharina, *Chorda filum* and dense red seaweeds on shallow unstable infralittoral boulders or cobbles (MIR.LsacChoR)	The community is composed of predominantly opportunistic and fast colonizing species. For instance, Kain (1975) recorded that *Laminaria saccharina* was abundant six months after substratum was cleared. Although the community might look very similar one year after loss of species, some species such as encrusting coralline algae that survive winter storms will not reach their previous extent on cobbles for some years. Recoverability from impact is therefore described as high.
Alcyonium digitatum with a bryozoan, hydroid and ascidian turf on moderately exposed vertical infralittoral rock (IR.AlcByH)	Many of the species in this and similar biotopes are fast colonizing and almost all sessile species have planktonic larvae or propagules. For instance, the likely initial colonizing species *Balanus crenatus*, heavily colonized a site that was dredged for gravel within 7 months (Kenny & Rees, 1994). Other species such as erect Bryozoa and Hydrozoa will settle on the barnacles and overgrow them. A 'qualitative climax' community was described as being reached within 26 months in the study of establishment on settlement panels in similar communities described by Castric (1977). Some species might be slower to colonize and grow, such as *Alcyonium digitatum*, but it is expected that close to a full complement of species will have re-settled within five years. However, a minority of species that live in the biotope may be less fast to settle and may be of marine natural heritage importance (for instance, *Hoplangia durotrix*). However, overall recovery is predicted to be high.

Appendix 6. List of represented and representative biotopes (see text). Researched biotopes are shown in bold.

LITTORAL ROCK (and other hard substrata)

LICHENS AND ALGAL CRUSTS

Biotope name		Biotope code
Yellow and grey lichens on supralittoral rock.		**LR.L.YG**
Represents	*Prasiola stipitata* on nitrate-enriched supralittoral or littoral fringe rock.	LR.L.Pra
	Verrucaria maura on littoral fringe rock.	LR.L.Ver
	Verrucaria maura and *Porphyra umbilicalis* on very exposed littoral fringe rock.	LR.L.Ver.Por
	Verrucaria maura and sparse barnacles on exposed littoral fringe rock.	LR.L.Ver.B
	Verrucaria maura on moderately exposed to very sheltered upper littoral fringe rock.	LR.L.Ver.Ver
Chrysophyceae on vertical upper littoral fringe soft rock.		**LR.L.Chr**
Represents	*Blidingia* spp. on vertical littoral fringe soft rock.	LRL.L.Bli
	Ulothrix flacca and *Urospora* spp. on freshwater-influenced vertical littoral fringe soft rock.	LR.L.Ulo.Uro

EXPOSED LITTORAL ROCK (mussel and barnacle shores)

Biotope name		Biotope code
***Mytilus edulis* and barnacles on very exposed eulittoral rock.**		**ELR.MB.MytB**
Barnacles and *Patella* spp. on exposed or moderately exposed, or vertical sheltered eulittoral rock.		**ELR.MB.Bpat**
Represents	*Chthamalus* spp. on exposed upper eulittoral rock.	ELR.MB.Bpat.Cht
	Barnacles and *Lichina pygmaea* on steep exposed upper eulittoral rock.	ELR.MB.Bpat.Lic
	Catenella caespitosa on overhanging, or shaded vertical, upper eulittoral rock.	ELR.MB.Bpat.Cat
	Barnacles, *Patella* spp. and *Fucus vesiculosus* f. *linearis* on exposed eulittoral rock.	ELR.MB.Fvesl
	Semibalanus balanoides on exposed or moderately exposed, or vertical sheltered, eulittoral rock.	ELR.MB.Bpat.Sem
***Corallina officinalis* on very exposed lower eulittoral rock.**		**ELR.FR.Coff**
***Himanthalia elongata* and red seaweeds on exposed lower eulittoral rock.**		**ELR.FR.Him**
Represents	Mixed red seaweeds on moderately exposed lower eulittoral rock.	MLR.R.XR
	Palmaria palmata on very to moderately exposed lower eulittoral rock.	MLR.R.Pal
	Mastocarpus stellatus and *Chondrus crispus* on very to moderately exposed lower eulittoral rock.	MLR.R.Mas

Biotope name		Biotope code
	Osmundea (Laurencia) pinnatifida and *Gelidium pusillum* on moderately exposed mid eulittoral rock.	MLR.R.Osm

MODERATELY EXPOSED LITTORAL ROCK (barnacle and fucoid shores)

Biotope name		Biotope code
Barnacles and fucoids (moderately exposed shores).		**MLR.BF**
Represents	*Pelvetia canaliculata* and barnacles on moderately exposed littoral fringe rock.	MLR.BF.PelB
	Fucus vesiculosus and barnacle mosaics on moderately exposed mid eulittoral rock.	MLR.BF.FvesB
	Fucus serratus on moderately exposed lower eulittoral rock.	MLR.BF.Fser
	Fucus serratus and red seaweeds on moderately exposed lower eulittoral rock.	MLR.BF.Fser.R
	Dense *Fucus serratus* on moderately exposed to very sheltered lower eulittoral rock.	MLR.BF.Fser.Fser
	Fucus serratus and piddocks on lower eulittoral soft rock.	MLR.BF.Fser.Pid
	Pelvetia canaliculata on sheltered littoral fringe rock.	SLR.F.Pel
	Fucus spiralis on moderately exposed to very sheltered upper eulittoral rock.	SLR.F.Fspi
	Fucus vesiculosus on sheltered mid eulittoral rock.	SLR.Fves
***Fucus serratus* and under-boulder fauna on lower eulittoral boulders.**		**MLR.BF.Fser.Fser.Bo**
Represents	*Laminaria digitata* and under-boulder fauna on sublittoral fringe boulders.	MIR.KR.Ldig.Ldig.Bo
***Enteromorpha* spp. on freshwater influenced or unstable upper eulittoral rock.**		**MLR.Eph.Ent**
Represents	*Porphyra purpurea* or *Enteromorpha* spp. on sand-scoured mid to lower eulittoral rock.	MLR.Eph.EntPor
	Ephemeral green and red seaweeds on variable salinity or disturbed eulittoral mixed substrata.	SLR.FX.EphX
***Rhodothamniella floridula* on sand-scoured lower eulittoral rock.**		**MLR.Eph.Rho**
***Mytilus edulis* and *Fucus vesiculosus* on moderately exposed mid-eulittoral rock).**		**MLR.MF.MytFves**
Represents:	*Mytilus edulis* beds on eulittoral mixed substrata.	SLR.Mx.MytX
	Mytilus edulis, *Fucus serratus* and red seaweeds on moderately exposed lower eulittoral rock.	MLR.MF.MytFR
***Sabellaria alveolata* reefs on sand-abraded eulittoral rock.**		**MLR.Sab.Salv**

SHELTERED LITTORAL ROCK (fucoid shores)

Biotope name		Biotope code
***Ascophyllum nodosum* on very sheltered mid eulittoral rock.**		**SLR.F.Asc**
Represents	*Ascophyllum nodosum* on full salinity mid eulittoral rock.	SLR.F.Asc.Asc
***Fucus ceranoides* on reduced salinity eulittoral rock.**		**SLR.F.Fcer**
Represents	*Fucus ceranoides* on reduced salinity mixed substrata.	SLR.FX.FcerX
Barnacles and *Littorina littorea* on unstable eulittoral mixed substrata.		**SLR.FX.BLlit**
***Fucus vesiculosus* on mid eulittoral mixed substrata.**		**SLR.FX.FvesX**
Represents	*Ascophyllum nodosum* on mid eulittoral mixed substrata.	SLR.FX.AscX
	Fucus serratus on lower eulittoral mixed substrata.	SLR.FX.FserX

LITTORAL ROCK (other)

Biotope name		Biotope code
Green seaweeds (*Enteromorpha* spp. and *Cladophora* spp.) in upper shore rockpools.		**LR.Rkp.G**
***Corallina officinalis* and coralline crusts in shallow eulittoral rockpools.**		**LR.Rkp.Cor**
Represents	Coralline crusts and *Paracentrotus lividus* in shallow eulittoral rockpools.	LR.Rkp.Cor.Par
	Bifurcaria birfurcata in shallow eulittoral rockpools.	LR.Rkp.Cor.Bif
	Cystoseira spp. in shallow eulittoral rockpools.	LR.Rkp.Cor.Cys
Fucoids and kelps in deep eulittoral rockpools.		**LR.Rkp.FK**
Seaweeds in sediment (sand or gravel)-floored eulittoral rockpools.		**LR.Rkp.SwSed**
Hydroids, ephemeral seaweeds and *Littorina littorea* in shallow eulittoral mixed substrata pools.		**LR.Rkp.H**
Overhangs and caves.		**LR.Ov**
Represents	Sponges and shade tolerant red seaweeds on overhanging lower shore bedrock.	LR.Ov.SR[3]
	Sponges, bryozoans and ascidians on deeply overhanging lower shore bedrock.	LR.Ov.SByAs[4]
***Rhodothamniella floridula* in littoral fringe soft rock caves.**		**LR.Ov.RhoCv**
Audouinella purpurea and *Cladophora rupestris* on upper to mid-shore cave walls		LR.FLR.CvOv.AudCla[5]

[3] Equivalent to biotope (version 04.05) 'Sponges and shade-tolerant red seaweeds on overhanging lower eulittoral bedrock and in cave entrances' (LR.FLR.CvOv.SpR) (Connor *et al.,* 2004).

[4] Equivalent to biotope (version 04.05) 'Sponges, bryozoans and ascidians on deeply overhanging lower shore bedrock or caves' (LR.FLR.CvOv.SpByAs)) (Connor *et al.,* 2004).

[5] Biotope classification 04.05 version codes (Connor *et al.*, 2004) with NO equivalent 97.06 version codes (Connor *et al.*, 1997).

Biotope name	Biotope code
Verrucaria mucosa and/or *Hildenbrandia rubra* on upper to mid shore cave walls	LR.FLR.CvOv.VmucHil[3]
Sponges, shade-tolerant red seaweeds and *Dendrodoa grossularia* on wave-surged overhanging lower eulittoral bedrock and caves	LR.FLR.CvOv.SpR.Den[3]
Faunal crusts on littoral wave-surged cave walls.	**LR.FLR.CvOv.FaCr[3]**
Sparse fauna (barnacles and spirorbids) on sand/pebble-scoured rock in upper littoral to lower shore caves	LR.FLR.CvOv.ScrFa[3]
Barren and/or boulder-scoured littoral cave walls and floors	LR.FLR.CvOv.BarCv[3]

INFRALITTORAL ROCK (and other hard substrata)

EXPOSED INFRALITTORAL ROCK

<table>
<tr><th colspan="2">Biotope name</th><th>Biotope code</th></tr>
<tr><td colspan="2">Alaria esculenta on exposed sublittoral fringe rock.</td><td>EIR.KfaR.Ala</td></tr>
<tr><td>Represents</td><td>Alaria esculenta, Mytilus edulis and coralline crusts on very exposed sublittoral fringe bedrock.</td><td>EIR.KfaR.Ala.Myt</td></tr>
<tr><td></td><td>Alaria esculenta and Laminaria digitata on exposed sublittoral fringe bedrock.</td><td>EIR.KfaR.Ala.Ldig</td></tr>
<tr><td colspan="2">Laminaria hyperborea forest with a faunal cushion (sponges and polyclinids) and foliose red seaweeds on very exposed infralittoral rock.</td><td>EIR.KfaR.LhypFa</td></tr>
<tr><td colspan="2">Laminaria hyperborea with dense foliose red seaweeds on exposed infralittoral rock.</td><td>EIR.KfaR.LhypR</td></tr>
<tr><td>Represents</td><td>Laminaria hyperborea forest with dense foliose red seaweeds on moderately exposed upper infralittoral rock.</td><td>MIR.KR.Lhyp.Ft</td></tr>
<tr><td colspan="2">Foliose red seaweeds on exposed or moderately exposed lower infralittoral rock.</td><td>EIR.KfaR.FoR</td></tr>
<tr><td>Represents</td><td>Foliose red seaweeds with dense Dictyota dichotoma and/or Dictyopteris membranacea on exposed lower infralittoral rock.</td><td>EIR.KfaR.FoR.Dic</td></tr>
<tr><td></td><td>Foliose seaweeds and coralline crusts in surge gully entrances.</td><td>EIR.SG.FoSwCC</td></tr>
<tr><td colspan="2">Sponge crusts and anemones on wave-surged vertical infralittoral rock.</td><td>EIR.SG.SCAn</td></tr>
<tr><td>Represents</td><td>Sponge crusts and anemones and Tubularia indivisa in shallow infralittoral surge gullies.</td><td>EIR.SG.SCAn.Tub</td></tr>
<tr><td></td><td>Sponge crusts and ascidians on wave-surged vertical infralittoral rock.</td><td>EIR.SG.SCAs</td></tr>
<tr><td></td><td>Dendrodoa grossularia and Clathrina coriacea on wave-surged vertical infralittoral rock.</td><td>EIR.SG.SCAs.DenCla</td></tr>
<tr><td></td><td>Sponge crusts, colonial (polyclinid) ascidians and a bryozoan/hydrozoan turf on wave-surged vertical or overhanging infralittoral rock.</td><td>EIR.SG.SCAs.ByH</td></tr>
<tr><td></td><td>Sponge crusts on extremely wave-surged infralittoral cave or gulley walls.</td><td>EIR.SG.SC</td></tr>
</table>

MODERATELY EXPOSED INFRALITTORAL ROCK

Biotope name		Biotope code
***Laminaria digitata* on moderately exposed sublittoral fringe rock.**		**MIR.KR.Ldig.Ldig**
***Laminaria digitata* and piddocks on sublittoral fringe soft rock.**		**MIR.KR.Ldig.Pid**
***Laminaria saccharina*, *Chorda filum* and dense red seaweeds on shallow unstable infralittoral boulders and cobbles.**		**MIR.SedK.LsacChoR**
Represents	Ephemeral red seaweeds and kelps on tide-swept mobile infralittoral cobbles.	MIR.SedK.EphR
	Mixed kelps with scour-tolerant and opportunistic foliose red seaweeds on scoured or sand-covered infralittoral rock.	MIR.SedK.XKscrR
	Saccorhiza polyschides and other opportunistic kelps on disturbed upper infralittoral rock.	MIR.SedK.Sac

INFRALITTORAL ROCK (other)

Biotope name	Biotope code
***Alcyonium digitatum* and a bryozoan, hydroid and ascidian turf on moderately exposed vertical infralittoral rock.**	**IR.FaSwV.AlcByH**