MARINE AGGREGATE SITE RESTORATION AND ENHANCEMENT

STRATEGIC POLICY OVERVIEW

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Prepared by Emu Ltd. on behalf of The British Marine Aggregates Producers Association, The Crown Estate and English Nature.

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Note from the Editors

This document has been prepared by Emu Limited for the British Marine Aggregate Producers Association, The Crown Estate and English Nature with the aim of stimulating debate on the feasibility and merits of undertaking restoration at marine aggregate dredging sites. It is, by intention, a discussion document and not a policy document and does not set out to be prescriptive in the methods and criteria for undertaking restoration in the marine environment although recommendations on approaches to the assessment of when, where and how to restore sites form a central theme.

The authors are grateful for the input and comments provided by representatives of the aggregates industry, the Crown Estate and English Nature who have helped to shape and inform the output. However, it is important to stress that the views expressed in this report are independent and do not represent the official views or policy of the British Marine Aggregates Producers Association, The Crown Estate, or English Nature.

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Executive Summary

Introduction

Partnership between the quarrying industry and conservation groups has been one of the success stories for sustainable development within recent years. Schemes such as the Needingworth Quarry and Wetland Project, which will generate 30 million tonnes of aggregate over a ten year period and ultimately create 700 hectares of wetland, are the culmination of several decades of convergence between the objectives of the quarrying industry and those directly entrusted with safeguarding nature conservation interests.

Needingworth Quarry
Source: Minerals & Nature Conservation Forum

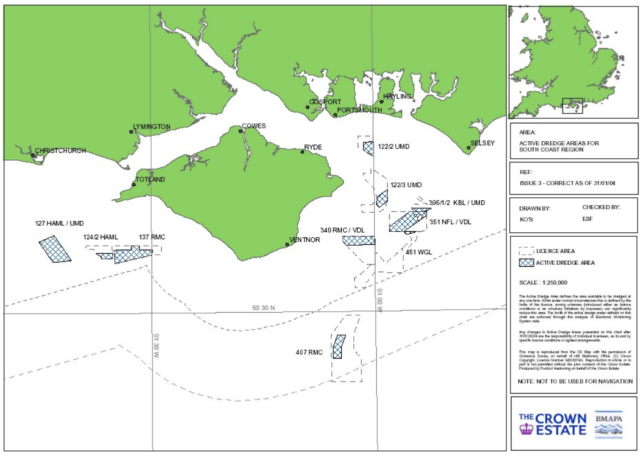
Given the close co-operation and understanding which now exists between the aggregates extraction industry and conservation groups, it is a natural progression to extend consideration of nature conservation objectives to offshore aggregate extraction sites. With nature conservation for the marine environment still in its formative stage, an exciting opportunity exists for BMAPA and The Crown Estate in partnership with English Nature to explore the issues associated with marine aggregate site remediation and to consider what might be achieved.

This study has the following objectives:

- **To review how the effectiveness of remediation at land based extraction sites can aid policy development regarding marine aggregate sites.**
- **To review the existing knowledge of the impacts of marine dredging on the physical and biological nature of the seabed, within and adjacent to the dredged area.**
- **To review existing methods and approaches to coastal and marine habitat remediation, and identify potential applications for marine aggregate dredging sites.**
- **To review the need for site rehabilitation at marine sites vs. the natural recovery process and the realistic applicability of the possible approaches to remediation identified.**
- **To identify the key points for discussion arising from the strategic review, in relation to the environmental and economic cost of the various rehabilitation options vs. the "do-nothing" option of natural site recovery.**
- **To identify gaps in the existing knowledge of dredging impacts, recovery and site remediation methods and, where appropriate, suggest research projects that will address these shortfalls and allow the suitability and feasibility of remediation methods to be scientifically assessed.**

Data has been collated on the impacts of dredging on benthic communities and the knock on effects on recovery from studies which encompass a variety of aggregate sites. Material has also been drawn from the terrestrial quarrying environment, the intertidal and coastal environment, and other marine industries. Representative studies have been selected to provide examples of both good and bad practice, which can form the basis for a set of guiding principles to apply to marine remediation.

Conservation Legislation and its Significance to the Aggregates Industry



Zoning for the South Coast Licence Areas
Source: BMAPA

The study briefly reviews key instruments of environmental legislation which govern the industry. Within this legislative framework, a comprehensive set of controls to minimise the impacts of the industry in the marine environment exists in the form of the Interim **Government View (GV) Procedure**. Since these measures have a direct bearing on impacts (and therefore recovery rates) they are pivotal to any debate on the place for remediation at marine aggregate sites.

Impacts of Dredging on Benthic Communities and on Recovery Rates

Dredging clearly has an impact on benthic communities but the nature of this impact varies widely according to the intensity of dredging and pre-existing environmental conditions - in particular the inherent stability or mobility of the seabed sediments. Recovery tends to be more rapid in unstable dynamic environments such as **shallow water mobile sands** typically ranging from a few months to between 2-4 years. Conversely in **deep water stable gravels** recovery of some long- lived species can take in excess of 15 years. While broad generalisations can be made on recovery in relation to sediment types, the considerable variation in recovery within and between habitat types dictate that meaningful assessments of recovery can only be undertaken on a site-specific basis, incorporating local environmental factors.

Electronic Monitoring System (EMS)
Source: The Crown Estate

Guiding Principles for Remediation

For the purposes of this study remediation has been defined as **the action taken at a site following anthropogenic disturbance, to restore or enhance its ecological value**. This can be achieved by a range of approaches from non-intervention through to habitat enhancement or creation. The guiding principles for remediation schemes set out below are suggested as a generic approach to remediation of marine aggregate sites.

1. **Establish the need for remediation** □ Government policy guidelines for remediation of marine aggregate sites are a good starting point to assess the need for remediation. In the absence of clear regulatory guidance, the need for remediation at existing sites needs to be determined on a case by case basis, drawing upon the best scientific knowledge available.

2. **Establish the goals and objectives for remediation of a site** – Current policy requires the seabed to be left in a similar condition to the pre-dredged state. This would suggest the goal of remediation should be restoration, as opposed to habitat creation or enhancement. Given the dynamic nature of ecosystems, 100% restoration of species composition and/or population may not be realistic, and there exists a need to define 'restoration' in a manner which can be generically applied to marine remediation.

3. **Determine the criteria against which the performance of a scheme will be measured** Since remediation, by definition, involves intervention to enhance the ecological value of a site, there has to be criteria against which one can measure the performance of a scheme. Without this, there is no means of establishing if the objectives are being achieved and intervention would be irresponsible.

4. **Adopt an ecosystem approach** - There is a need to draw a conceptual boundary around the ecosystem under consideration within any remediation scheme – but this is particularly true of the marine environment which is an inherently open and dynamic system. Scientific knowledge can guide the process, but decision makers need to limit the natural and anthropogenic factors to be taken into consideration.

5. **Follow an Environmental Impact Assessment (EIA) process** - All environmental factors need to be taken into consideration, and decisions made based upon the significance of both the positive and negative impacts. Collation of baseline environmental data is essential to determine both the need for remediation and to identify the preferred method and its associated impacts. Pre, in-progress and post remediation monitoring regimes will allow progress of the scheme to be evaluated.

Remediation at marine aggregate dredging sites is very much a pioneer technology and the long-term effectiveness of different techniques in most instances has yet to be evaluated. Experience with terrestrial and coastal remediation projects can inform our approach to remediation within the marine environment. However, the transfer of knowledge is predominantly in terms of principles which can be applied rather than actual techniques which have little application given the fundamentally different nature of the environment and different scale on which excavation is undertaken.

Remediation Methods with Potential Application to Marine Aggregate Sites

The following suggested approaches to remediation of marine aggregates sites are based on techniques used in a variety of marine and coastal applications, but also draw upon key principles from the quarrying industry.

- **Non Intervention / Natural Recolonisation** Where intervention in terms of remediation is not likely to have a significant impact on the rate of natural recovery, a non intervention approach may be appropriate. In conjunction with monitoring, non intervention and natural recolonisation is the current management option for most UK dredging sites.

- **Active-Passive Recovery** allows natural recolonisation to take its course, with the additional measure of designating a non-disturbance zone / marine protected area (MPA) around a site. This 'buffer' zone around a dredged site aims to improve the natural rate of recovery by excluding seabed disturbances and therefore has particular application to naturally stable areas.

- **Conserving the Modified Seabed** Evidence suggests that dredging activity, particularly that associated with static dredging, can sometimes create a distinctly different habitat from the surrounding area, in which the changes to the seabed topography can provide micro-niches for certain species which are of enhanced biodiversity value. The conservation of the now modified seabed may be regarded as a remediation approach.

Pentapora folicacea (Ross Coral)
Source: Seasearch

- **Marine Site Restoration** While site restoration is extensively practiced on land, subtidal restoration in the UK appears to be very much in an early experimental stage. A common theme for all of the existing examples is the process of **capping**, covering an impacted area of seabed with a preferred sediment layer, to entrap contaminated material and/or to encourage the recovery of a more natural biotope. To be effective it involves considerable amounts of material to be deployed over a small area. Limitations of scale and cost are foremost in considering physical restoration schemes. However, there is merit in exploring the beneficial use of some waste materials in this context e.g. rejected aggregate dredgings or processed scallop shells. Site restoration may be appropriate for targeted areas which are ecologically impoverished or where the sediment composition has changed radically due to dredging.

- **Marine Habitat Creation / Enhancement** Differentiation between habitat restoration and habitat creation / enhancement is not clear cut and is more a factor of management objectives than methods. A range of existing examples have been identified which fall into this broad category of marine environmental management, including the seeding of gravel or shells in order to improve the seabed substrate (notably for shellfish harvesting areas) and the use of artificial reefs and fish aggregating devices. The approach may be used either to promote habitat features broadly consistent with adjacent areas or enhance the habitat diversity of an area in order to increase productivity and/or biodiversity. The latter principally focuses on the use of artificial structures. Both of these aims could be applicable to the management of aggregate dredging areas, but both also have potential practical, political, and localised environmental issues associated with them. As above, the application of this approach is probably highly specific for ecologically impoverished areas, possibly as a component within wider biodiversity plans.

Assessing the Need for Remediation

This study has sought to address the question: where is remediation appropriate; i.e. for which habitats, in response to which dredging conditions and judged by which criteria?

An approach is needed which formalises and standardises assessment of impact and recovery rates across all sites while assimilating the site specific environmental factors which control these processes. Formalised risk assessments such as that developed by CEFAS for evaluating the impacts of dredging operation on fisheries issues have merit in achieving greater standardisation and objectivity in data collation, while incorporating informed scientific judgement in the assessment process. A similar model might be used to assess the impacts of dredging on benthic communities.

Government policy makes provision for 'corrective action' (i.e. remediation) where monitoring of dredging activity identifies unacceptable impacts on the marine environment. What is currently absent is a robust 'trigger mechanism' for initiating remediation.

Benthic Sampling Using a Hamon Grab
Source: Emu Ltd.

Parallel sampling at treatment (aggregate) sites and reference sites, with rigorous quality assurance, would appear to offer a plausible means of measuring change from a baseline condition. In this respect it could be used to monitor the process of natural recovery, following dredging. Where natural recovery falls significantly behind expectations (based upon pre-determined site specific ecological quality objectives (EcoQOs) it could serve as a trigger for appropriate intervention. The success or otherwise of remediation in aiding recovery could then be judged by the same criteria.

Potentially, this procedure could form part of the 'aftercare' considerations within a dredging licence application.

Dredging clearly has impacts on the environment but remediation can also have impacts and involves significant commitment in finance and other resources. Furthermore, remediation in the marine environment is a relatively untried process in the UK. The costs of remediation have to be weighed up against the ecological benefits derived.

The Future of Remediation within the Marine Aggregate Industry

In determining the place for remediation in the marine aggregate industry, the role of the regulator is pivotal. The detailed requirements embodied in the government's precautionary approach to licensing and monitoring should form the basis for a thorough consideration of remediation, taking into account all of the issues associated with this potentially complex process.

Currently, the powers of the Secretary of State to intervene where he or she considers that restoration objectives post-dredging will not be met, are not fully executed because there is no objective, agreed, effective trigger mechanism in place at most sites.

Development of an appropriate mechanism for restorative action is considered an essential step in remediation policy for marine aggregate sites and one that can only be achieved through broad consensus of all parties concerned. Such a system should guard against indiscriminate regulatory measures governing mitigation that could be cost prohibitive for developers or non-beneficial for the environment.

Recommendations

- **Establish a marine dredged site management working group involving industry, regulators, key stakeholders and technical experts.**

- **Consider priorities for further targeted research into dredging impact and recovery rates, specifically related to a variety of dredging intensities in the most sensitive habitats (i.e. stable gravels).**

- **Develop robust criteria for assessing recovery and determining when intervention in the form of remediation is necessary.**

- **Explore the contribution which the aggregates industry can make to marine BAPs and SAPs and to participation in site management of future offshore Natura 2000 sites.**

Dredger in Operation
Source: BMAPA

Contents

1.0 Habitat Restoration / Enhancement at Marine Aggregate Sites: Opening the Debate

1.1 Introduction

The perception of conservation within the UK minerals industry has changed radically over the past three decades. Making provision for restoration of the visual landscape, flora and fauna which was formerly an encumbrance to mainstream business, or at best a distraction, is now a fully integrated goal at the licence application stage. This transition has occurred as opportunities for joint action have been grasped by far-sighted industry representatives and conservationists in sustainable development initiatives, many of which predate Gro Harlem Bruntland's coinage of the phrase in "Our Common Future" (1987). The success of integrating conservation interests within the minerals industry are plain to see with over 700 former extraction sites now designated as sites of special scientific interest (SSSIs).

More can always be achieved and success breeds its own momentum. It is understandable, therefore, that the industry and conservation groups should wish to extend consideration of conservation objectives into areas which have received comparatively less attention. It is in this context that the present study has been conceived recognising also the strategic importance of marine aggregates and the increasing focus demanded by regulatory authorities for stewardship of the marine environment.

The feasibility or indeed the desirability of undertaking habitat restoration or enhancement at marine sites has not been fully investigated. This report has been specifically commissioned by BMAPA, the trade association for the marine aggregates industry, and The Crown Estate, the custodian of the near-shore seabed, in partnership with English Nature, the statutory adviser on nature conservation in England, to explore the issues associated with marine aggregate site remediation and to stimulate discussion and debate on the key issues. Brief details of the lead organisations are given below.

Questions regarding remediation and aftercare of some dredge sites are increasingly being raised during EIA consultation for new licence applications, and a focused treatment of the subject is therefore timely.

This study aims to provide the first step in that process, through a review of the existing knowledge of the impacts of dredging on the physical and biological nature of the seabed (within and adjacent to marine dredging areas), and the need for consideration of further remedial action.

No attempt is made to produce prescriptive guidance on the subject of remediation of marine aggregate dredging sites at this stage, but rather we have set out to provide an informed view of the subject which will stimulate discussion across the industry, regulators and the interested stakeholders.

It is expected that subsequent projects will be required to develop more specific guidance on when, where, and how particular methods of rehabilitation may be applied. Some areas for further study are offered as an output from this study.

1.1.1 BMAPA

The British Marine Aggregate Producers Association (BMAPA) was formed in 1992, and comprises members of the Quarry Products Association with marine aggregate dredging interests. The UK operates around 35 dredging vessels, on 78 production licenses around the UK. The vessels are almost entirely British registered and carry British crew. BMAPA represents the industry and its members are: Britannia Aggregates Ltd, British Dredging Ltd, DEME Building Materials Ltd., Hanson Aggregates Marine Ltd, Kendall Bros. (Portsmouth) Ltd, Northwood (Fareham) Ltd, Norwest Sand and Ballast Ltd, RMC Marine Ltd, United Marine Dredging Ltd. and Volker Dredging Ltd.

1.1.2 The Crown Estate

The Crown Estate dates back to the reign of King Edward the Confessor and is part of the hereditary possessions of the Sovereign "in right of the Crown". The estate is managed under the provisions of The Crown Estate Act 1961 by the Crown Estate Commissioners, who have a duty to maintain and enhance the value of the Estate and the income derived. The net revenue surplus is paid to the Exchequer.

The Crown Estate includes;

- More than 120,000 hectares of agricultural land in England, Scotland and Wales,
- Substantial blocks of commercial property, primarily in London,
- An extensive marine estate covering 55% of the foreshore and most of the seabed out to the 12nm Territorial Limit
- Mineral rights throughout to UK's territorial limits.

The Crown Estate issues licenses on behalf of DEFRA for marine aggregate dredging and are responsible for overseeing and monitoring the effects of dredging. Currently, 72 licenses are issued and there are 29 production licence applications in the pipeline for new dredging areas. Revenue from these licenses constitutes the largest single source of revenue for the Marine Estate, contributing at present around £13.8m / year (40 -50% Marine Estate revenue).

1.1.3 English Nature

English Nature champions the conservation of wildlife and geology throughout England. It is a government agency set up under the Environment Protection Act 1990 and funded by the Department of Environment, Food and Rural Affairs. English Nature is currently developing a maritime strategy to help implement the Government's stewardship vision set out in 'Safeguarding our Seas'.

1.2 Some Definitions

At the outset of this study, it is helpful to provide definitions of some of the key terms and concepts which will be discussed.

Aftercare

The term 'Aftercare Condition' currently has a statutory definition in schedule 5 of the Town and Country Planning Act 1990 and schedule 3 of the Town and Country Planning (Scotland) Act 1997 which can be found in Annex C2.
There are currently no equivalent aftercare conditions attached to marine aggregate production licenses excepting the stipulation that:

Dredging should aim to leave the seabed in a similar physical condition to that present before dredging started in order to enhance the possibility of, and rate at which, the seabed recovers physically and biologically to its pre-dredging condition.

It would be premature to provide a definition for Aftercare Condition as it should apply to the marine environment. The concept of aftercare may be extended to the marine environment which we would define as **operations undertaken, post dredging, to bring the seabed up to an agreed condition** in order to promote ecological recovery or enhancement. The criteria and process for determining an appropriate condition, post dredging is a complex but timely issue for debate and the stimulus for this study.

Mitigation

Mitigation is defined in EC Directive 85/337 (EIA Directive) as **'measures envisaged in order to avoid, reduce and, if possible remedy significant adverse effects'**. Mitigation is currently incorporated in licensing arrangements for marine aggregate dredging via a number of measures including restriction of the area to be dredged via zoning; restriction of the quantity of aggregate recovered; seasonal restrictions; and prohibitions on screening (i.e. the sifting of mixed gravel and sand to retain one commodity and return the other to the sea). A more detailed review of mitigation measures can be found at 2.5.2.

Recovery

The process of recovery following environmental disturbance is generally defined as **the establishment of a community that is similar in species composition, population density and biomass to that previously present or at non-impacted sites**. (C-CORE 1996 as cited by Newell 2002).

Remediation

For the purposes of this study the core process of remediation is taken simply as '**the action taken at a site following anthropogenic disturbance to restore or enhance its ecological value**'. This broad inclusive definition has been selected to embrace all measures, both physical and administrative, which can be used to improve the ecology of a site and will be developed further in chapter 5.

1.3 Objectives

This desk based study has set out to:

- Explore how experience with aftercare at land based extraction sites (the success of which has been a driver for the present initiative) can inform policy development for marine aggregate sites.

- Review the existing knowledge on the impacts of marine dredging on the physical and biological nature of the seabed, within and adjacent to the dredged areas. A review of studies on the natural recoverability of the seabed is included. Due to the constraints of available literature, this study concentrates on the review of benthic impacts and recoverability as the most effective and appropriate indicator of dredging impacts. It is acknowledged that wider ecosystem effects could occur and may equally benefit from mitigating measures including remediation techniques.

- Review existing methods and approaches to coastal and marine habitat remediation, and identify potential applications for marine aggregate dredging sites.

- Review the need for site rehabilitation at marine sites vs. the natural recovery process and the realistic applicability of the possible approaches to remediation identified.

- Identify the key points for discussion arising from the strategic review, in relation to the environmental and economic cost of the various rehabilitation options vs. the "do-nothing" option of natural site recovery.

- Identify gaps in the existing knowledge of dredging impacts, recovery and site remediation methods. Where appropriate, suggest research projects that will address these shortfalls, and allow the suitability and feasibility of remediation methods to be scientifically assessed.

1.4 Method

There were two principal lines of initial inquiry for this study:

1. Review of existing scientific studies into the impacts of dredging on the physical and biological nature of the seabed. The principal aim being to identify the type of criteria upon which to base remediation decisions.

2. Review of available literature on approaches to remediation, drawing appropriate examples from the terrestrial, coastal and marine environments.

In sections 5 and 6 of the report, the two strands are brought together to discuss the potential for the remediation options, identified, in the context of the impact scenarios. Key questions which remain unanswered are then explored, based on the 'Need' and 'Approaches' elements to provide the main output. This will also include, where possible, suggestions for policy targets and guiding principles for site remediation, together with suggestions for approaches which appear to provide promise for use at marine dredged sites. Prioritisation of these then form the basis for the final recommendations. A schematic of the process is shown in Figure 1.1.

1.5 Data Sources

The study aims to cover a spectrum of different aggregate dredging environments around the UK, taking account of differences in seabed types, water depth, geographic variation and benthic faunal diversity and productivity. Where possible, examples have been selected which have been subject to both intensive marine aggregate dredging and detailed impact monitoring studies. However in the case of the deep water stable gravels such as the Eastern English Channel which constitutes a very distinctive environment quite different from any existing dredging site, it has only been possible at present to speculate on the impacts of dredging.

In considering how remediation in the marine environment might be approached, further material has been drawn from the terrestrial quarrying environment, the intertidal and coastal environment, and other marine industries. Representative studies have been selected to provide examples of both good and bad practice, which can form the basis for a set of guiding principles to apply to marine remediation.

In addition, correspondence, interviews and meetings with remediation practitioners, industry representatives and conservation bodies have contributed to developing a sound overview of the subject area.

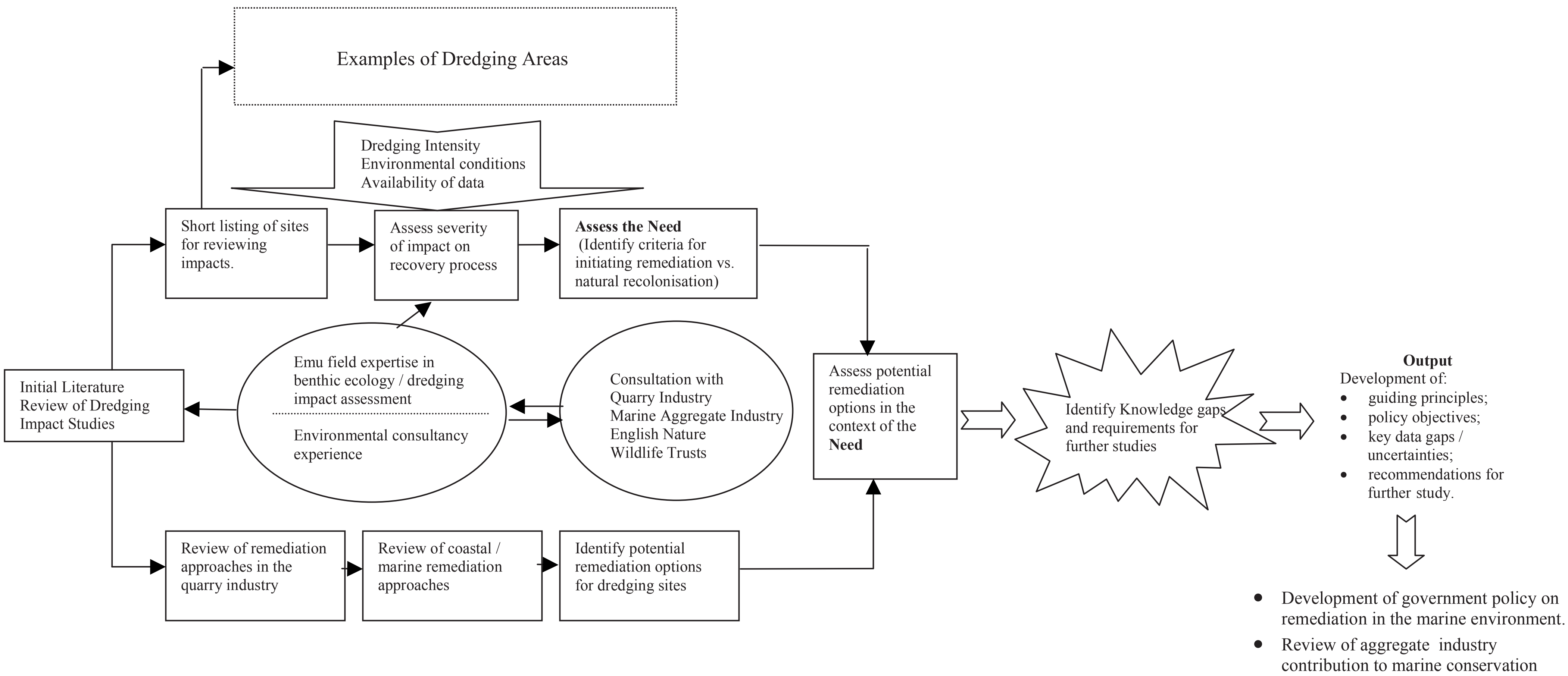


Figure 1.1: Marine aggregate site restoration and enhancement: schematic of the approach to the study.

2.0 Background to the Marine Aggregates Industry & Development of Conservation Policy

The past, present and future direction of the marine aggregates industry can best be understood by reference to the wider minerals industry, since there is overlap in terms of the market forces, management constraints and nature conservation values, affecting both sectors. The development of conservation interests within the quarrying industry, is reviewed in this chapter as a means of informing the approach, which the marine aggregate industry might take in relation to habitat remediation. The activities of both the quarrying industry and the marine aggregate industry are also briefly considered in relation to the UK Biodiversity Action Plan (UKBAP).

2.1 The Origins of the Marine Aggregate Industry

The origins of the UK marine sand and gravel industry can be traced back to the 1550's, where available aggregate from the River Thames constituted a cheap and plentiful supply of ballast for unladen sailing ships.

By the 17th Century, the provision of ballast from marine sources had developed into a major operation controlled by Trinity House, with the management of ballast becoming the organisation's principle source of income. This in turn provided the means to finance the nation's lighthouses, lightships and buoys.

The advent of modern aggregate dredging arguably dates from the 1930s, when the use of deck mounted pumps on vessels, in conjunction with suction pipes (primitive suction dredgers), greatly improved the efficiency of dredging. Suction dredging remains the industry standard today.

As with other mineral resources, the development of the marine aggregate industry has been shaped by the overall availability and demand for the resource nationally.

Figure 2.1 provides a simple overview of the surface geology for the United Kingdom. Broadly speaking, one can draw a line between Flamborough Head on the East coast and Portland in Dorset, with the surface geology to the north and west of this line predominantly comprising older and harder igneous and metamorphic rocks, whilst the area to the south and east is made up of sedimentary rocks (Russell 2003).

Where hard rock types are available, demand for coarser aggregates can be met from quarries. Consequently, this has been the traditional source of coarse aggregates in the north and west of England and Wales. But the absence of sedimentary rocks has resulted in a shortage of sand in these areas. Conversely for the land to the east of the imaginary line, the "soft" sedimentary rocks yield a natural source of sand from terrestrial sources, but a deficiency of coarser aggregates. Increasingly, however, competing demands for land use and the environmental constraints upon aggregate extraction from land sites is driving the search for alternative sources, including marine aggregates.

The current distribution and supply of marine aggregate extraction sites largely reflects the surface geology. Marine aggregate dredging to the west of England and Wales principally supplies sand, whilst to the east of England a mix of coarse aggregates and sand is landed in roughly equal proportions (M Russell (BMAPA) *pers. comm.*).

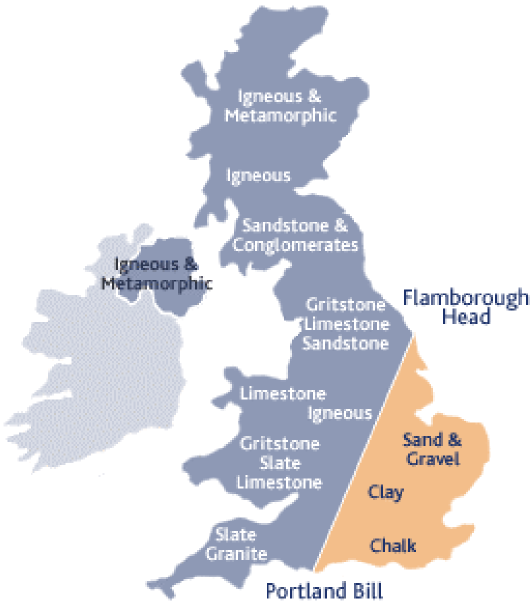


Figure 2.1: Simple Overview of the Surface Geology for the UK
(Source of the Quarry Products Association.)

2.2 The Market for Marine Aggregates

Aggregates are essential for development, required for every component of the construction industry. Every new house accounts for about 50 tonnes of aggregate and the per capita consumption amounts to over 4 tonnes per year (Russell 2003). The principal end uses for marine aggregates are shown in Figure 2.2. The UK construction industry, accounts for about 62% of consumption, predominantly as a constituent in concrete. Exports of marine aggregates, which accounted for 20% consumption in 1997, now represent 31% of market consumption. The remaining 7% is used in beach nourishment. This breakdown of market shares does, however, vary widely according to market demand.

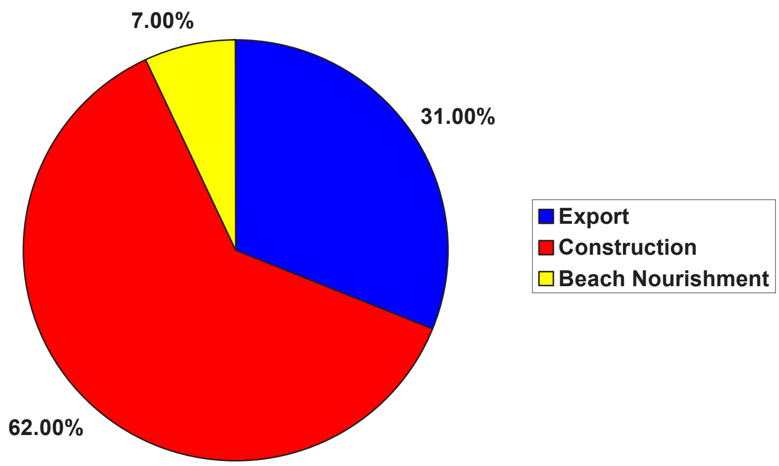


Figure 2.2: The Market for UK Marine Aggregates
(Source: Department of the Environment (1994) Minerals Planning Guidance Note 6: Planning Guidance for England and Wales.)

Figure 2.3 illustrates the regional variations in the market for marine aggregates. As a low-cost bulk material, transportation is a critical factor in determining the profitability of reserves. The concentration of marine aggregate extraction activity in the southern North Sea is the result of the location of the principal local markets for sand and gravel in the South East of England (particularly London), and near-continental Europe notably Holland and Belgium, coupled with the distribution of commercially viable reserves.

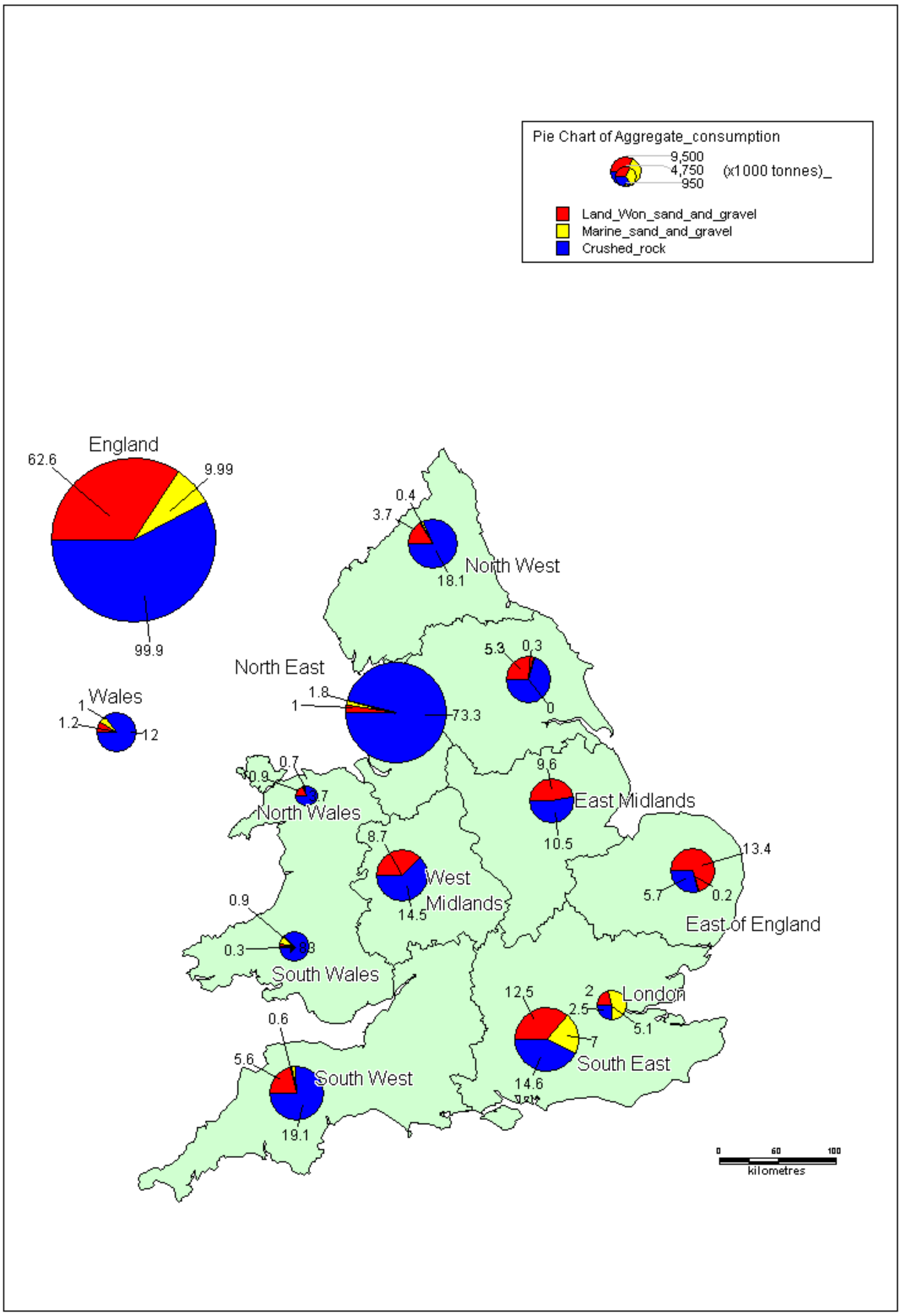


Figure 2.3: Consumption of Sand and Gravel and Crushed Rock for Aggregates 2001
(Source: Regional Dredging Statistics 2002, The Crown Estate)

The volume of marine sand and gravel extracted from the east coast, Thames and south coast regions amounts to about 17 million tonnes per year, of which 10 million is delivered to the UK. Marine sand & gravel is particularly important for London and the South East, where it accounts for well over a third of regional demand. The remaining 3-4 million tonnes of marine aggregate extracted in the UK is supplied to the more peripheral markets in the north east, the Bristol Channel and the Irish Sea.

A number of factors account for the contribution made by the marine aggregates industry to the overall aggregates supply over the past 4 decades.

- Overall demand for aggregates has increased markedly. Figure 2.4 shows that overall use of primary aggregates in England and Wales doubled over the period 1960-1991.
- Use of marine aggregates doubled during the 1970s, peaking at 27 million tonnes per year in 1987, and reducing to between 19 and 23 million tonnes in the period since. Year to year variations largely reflect the economic conditions.
- Land won sources have become increasingly scarce, due both to diminishing resources and to the constraints imposed in areas of high agricultural, environmental or development value, particularly in the South East.
- Changes in government policy, which have periodically reviewed the contribution which should be made by marine aggregates to meet demand.
- Use of larger vessels which have increased the economic viability of marine aggregate extraction allowing transport of cargo over greater distances
- Developments in sonar imaging and the advent of satellite navigation that have allowed the quantification of marine reserves to increasing degrees of accuracy and improved resource management.

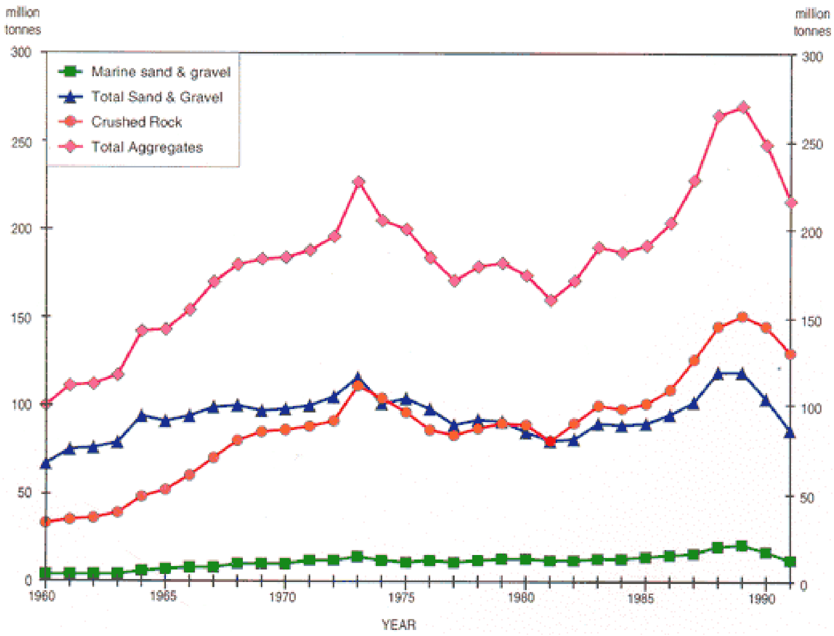


Figure 2.4: Primary Aggregates Consumption, England and Wales 1960-1991
(Source: Department of the Environment (1994) Minerals Planning Guidance Note 6:

Planning Guidance for England and Wales.)

2.2.1 Future Aggregate Demand and Implications for Marine Aggregates

Draft National and Regional Guidelines for Aggregate Provision in England over the period 2001 to 2016 were recently published by the Office of the Deputy Prime Minister (ODPM 2002). These forecast a total aggregate demand over the period of 3.4 billion tonnes, which represents an average consumption of 212.5 million tonnes per annum, a reduction of around 24% on the MPG6 1994 figure of 280 million tonnes per annum (Department of the Environment 1994). This reduction in forecast demand is a reflection of the lower production levels that have been experienced since 1994.

In terms of sources of supply, the draft guidelines propose a significant shift away from the provision of land-won reserves (67% as opposed to 63%). The non land-won supplies will be derived from secondary and recycled materials (24% as opposed to a previous 12%), net imports to England (4% as opposed to a previous 7%), and marine sand and gravel (5% as opposed to 7%).

With respect to marine won sources, the assumed provision is a total of 173 million tonnes over the period of the guidelines, which equates to 11 million tonnes per annum.

It is important to stress that the above draft regional guideline have been issued for consultation purposes only at this stage and the basis of the forecast and projected demand is strongly contested by the Quarry Products Association (QPA).

An independent study commissioned by the QPA and undertaken by the Centre for Economics and Business Research (CEBR) indicates the likelihood of a growth trend in demand for total aggregates from 212 million tonnes in 2001 to 261 million tonnes in 2016 (QPA 2003). This projection contrasts strongly with the government's relatively flat outlook for aggregates demand. The CEBR have criticised the government's figures which they consider are derived from an over-simplified projection of growth in the construction sector which fails to take into consideration government infrastructure development plans in particular.

Currently the industry view is that the government will revise its forecast demand for total aggregates upwards having taken into consideration the reports presented during consultation (personal communication McLaughlin (QPA) Jan 2004). If this view is accurate, the forecast for the UK market for marine aggregates is likely to be one of steady growth rather than a scenario of static or possibly declining demand predicted by the draft regional guidelines.

The projected increase in demand for marine aggregates is further substantiated if one takes into account the rapid growth in the export market since 1997. The future of that export market is difficult to predict, since it will be determined by the policies of the states concerned, which are regularly reviewed. Currently there are no restrictions placed on the volume of marine aggregates exported from the UK.

2.3 Scale of Marine Aggregate Operation Relative to the Land Based Sector

A line of investigation within this study is to examine whether conservation practice within the quarrying industry can inform an approach to habitat remediation for the marine aggregate industry, and if so, how? This line of inquiry, while helpful, needs to be set in the context of the relative scales at which extraction is undertaken within the land based and marine sectors. From Figure 2.4

above, it can be seen that marine-won aggregates overall has tended to provide little more than a quarter of the total share of sand and gravel.

	Land Won	Marine	
Total area licensed (sand & gravel)	**270 km^2**	**1300 km^2*[2]**	
Total area worked annually	**150 km^2 *[1]**	**150 km^2*[2]**	
% area worked annually	**55**	**11.5**	
Volume extracted annually (England & Wales)	**70 mta**	**23 mta**	
Max. size of site	**9.45 km^2 (Needingworth)**	Licence dredge area	**165km^2**
		Active dredge area	**106km^2**
Average size of site	**< 1km^2 *[1]**	Licence dredge area	**11.6km^2**
		Active dredge area	**24.7km^2**

*[1] Information supplied by QPA based on qualitative assessment.
*[2] BMAPA & The Crown Estate - The Area Involved – Fifth Annual Report (2004).

Table 2.1: Relative Sizes of Extraction Activity for Land and Marine Aggregate Sites

Table 2.1 shows the relative areas of land and seabed licensed and worked for aggregates on land and on the seabed, respectively.

This comparison clearly indicates that there are very significant and fundamental differences in the intensity and spatial scale of aggregate extraction at marine aggregate sites when compared with terrestrial sites.

This is a key point that needs to be borne in mind at the outset when drawing parallels between the approaches to remediation or restoration at terrestrial and marine aggregate sites and the development of possible approaches to restoring marine dredged sites.

This contrast can be summed up by the following two, key points:

- **The intensity of terrestrial aggregate extraction greatly exceeds that of marine aggregate extraction**. The volume extracted per unit area for land won sand and gravel is over five times that of marine aggregate sites. Typical extraction rates for sites in the Thames Valley are 50,000 tonnes / ha. per annum. By contrast the average extraction for marine sites is 1,476 tonnes / ha. per annum (based upon total production against area dredged. By way of further comparison, the volume extracted per unit area for hard rock quarries may be 50 times higher that for land gravel extraction sites, penetrating to around 100m below the surface.
- **The spatial extent of marine aggregate extraction is significantly greater than that for quarrying of aggregates**, albeit a much smaller percentage is actually worked in any one year. This in part reflects the policy of zoning, itself a measure introduced to minimise both the area of seabed impacted by dredging and the constraints placed upon the fishing industry.

The considerably higher intensity per unit area associated with terrestrial quarrying has significant long-term physical, visual and ecological impacts for which recovery, without human intervention, extends over very long timescales. The physical alteration to the profile of the seabed produced by marine aggregate trailer suction dredging is small by comparison to most land excavations and typically affects only the top few metres or so of the sediment (although this is dependent on the vertical nature of the deposit and can be deeper at some sites over time). When static dredging is employed this will tend to leave deeper depressions on the seabed typically more than 10m depth (depending upon the depth of the resource). Further details of dredging methods can be found at section 3.3.

Arguably, the impact of marine aggregate dredging while spatially more extensive tends to be less severe in the long term in relation to the eventual recovery of habitats and, in contrast with quarrying, generally affects only the upper, surface sediments. Furthermore, unlike land quarrying, marine dredging does not entail wholesale removal of material from a site but rather tends to leave patches of unaffected seabed which allows recolonisation of the dredged seabed to take place from these relatively undisturbed areas.

Principles which have been applied to the remediation of aggregate quarries on land will clearly, in many cases, have no direct application in the marine environment (e.g. flooding of sites to create wetland reserves). Where possible parallels exist, however, their application will need to consider the differences in scale and environmental conditions between these two different activities. Restoring the topography of the landscape of former extraction sites has been extensively used in the quarrying industry but to attempt physical restoration on the seabed for a licence area, discounting the merits / flaws in such a proposal, is unlikely to be feasible at every dredged site. However, topographical restoration of the seabed may be appropriate on a different scale for example at certain depressions resulting from static dredging.

Substantial variation in the scale and nature of marine aggregate dredging when compared with terrestrial quarrying and in the radically different nature of the impacted environments significantly limits the application of terrestrial remediation methods to marine extraction sites. This issue is considered in further detail in section 5.

2.4 The Marine Environment and Significance of Natural Stress

Different ecosystems experience differing degrees of natural stress. Ecological communities and the organisms which populate them tend to be able to adapt to changes due to physical, biological or chemical factors within any given environment which will tend to make them resilient to some degree of change and allow the ecosystem to remain in a relative state of constancy or **homeostasis**. These recovery mechanisms can play an important role in recovery following the impact of human activities such as dredging.

The resilience of communities to natural stress is thus an important factor in affecting recovery rate, particularly within the marine environment where dynamic physical stresses contrast strongly with the stability of most terrestrial environments in the UK.

The marine environment is a high energy environment, particularly in the coastal zone where the action of winds and waves affects the water column and the bottom sediments. The dynamic nature of sand and gravel environments, typical of many marine aggregate sites, has a strong bearing on the habitats and species which they support.

Highly mobile sand /gravel environments which experience continual but varying degrees of natural stress tend to be populated by opportunistic species in which high mortality rates are balanced by rapid life cycles and reproductive rates.

In comparatively more stable environments, for example deep water coarse gravels, such as are found in the Eastern English Channel, the lower degrees of natural stress favour equilibrium species in which organisms are selected for maximum competitive ability where space for settlement and growth is limiting. Longer life cycles and slower recruitment make such communities more susceptible to stress, be that from natural or anthropogenic sources.

In summary:

- **The dynamic physical stresses which operate in the marine environment are a significant factor affecting the resilience of communities to change and their capacity for recovery following disturbance**; there are few parallels to be found at terrestrial mineral extraction sites.

- **There is considerable variation in the exposure to natural stress within marine sand and gravel habitats;**

2.5 Nature Conservation, Biodiversity and the Aggregates Industry

The Minerals Industry as a whole has an established tradition in nature conservation, developed over more than three decades of experience. The land-based aggregates sector has particular expertise in habitat restoration, enhancement and creation. Quarrying, by its very nature, is a temporary activity and there is consequently a need to restore, or landscape sites where activity has ceased.

The key legislation which underpins nature conservation within the quarrying industry is detailed below:

- The Wildlife and Countryside Act 1981 as amended by Countryside and Rights of Way Act (2000).

- The European Community Directive on the conservation of Natural Habitats and of Wild Fauna and Flora (the Habitats Directive), and the European Community Directive on the conservation of wild birds (the Birds Directive). Both are enacted in the UK through The Conservation (Natural Habitats &c.) Regulations1994.

- The Environment Impact Assessment Directive (85/337 EEC as amended by EC Directive 97/11) Implemented in the UK by the Town and Country Planning Act 1990.

- Government policy is further reinforced in Planning Policy Guidance Note 9, Nature Conservation (PPG9).

The land based minerals industry makes a significant contribution to conservation, with approximately 700 Sites of Special Scientific Interest (SSSIs) having formerly been quarries, this equates to over 10% of SSSIs in the UK.

Former sand and gravel quarries have been created into open water habitats providing important areas for birds. In one wetlands study, quarries comprised 11-12% of the sites, and supported over 20% of the breeding water birds recorded; with much larger percentages of certain breeding populations recorded including great crested grebe and tufted duck (Owen et al 1986 as cited in English Nature et al 1999).

Figure 2.5: Great Crested Grebe
Source: Sussexbirder website.

While legislation has formed the bedrock for conservation policy, a proactive and responsible approach industry-wide, in partnership with English Nature, the Wildlife Trusts, the RSPB and a host of national and local conservation bodies, has seen a shift in position from statutory compliance to activism within the UK quarry industry.

In July 1988, the Quarry Products Association, the Silica and Moulding Sands Association (SAMSA) and English Nature signed a statement of intent, in which all three organisations committed to work together to achieve environmentally sustainable development within the minerals industry. In particular, they seek to foster communication and share examples of best practice with the wider community. The same partners have formed a Minerals Conservation Forum with its own website and promulgated a Biodiversity and Mineral Handbook to guide quarry operators in making best use of opportunities for habitat creation at terrestrial sites. In 2004, BMAPA, in response to an invitation, joined the forum.

There has been a recent change in strategy on the part of the quarrying industry, which is seeking to maximise its contribution to biodiversity, by aligning its conservation efforts more closely within the existing management structure for biodiversity.

In furtherance of its obligations under the Convention on Biological Diversity, signed at the 'Earth Summit' in 1992, the UK government published a Biodiversity Action Plan for the UK (UKBAP). This identifies the means by which the UK should contribute to biological conservation of biodiversity over the ensuing twenty years. Detailed plans have been prepared by the UK Biodiversity group, outlining action which can be taken to conserve and enhance existing habitats and species, including the setting of national targets.

Within the national plan, there is a network of local BAPs working to local plans and targets. By integrating its conservation efforts within the local BAPs, the quarrying industry ensures its contribution enhances the focused efforts undertaken by local authorities, voluntary conservation groups and government to promote biodiversity.

2.5.1 Nature Conservation, Biodiversity and the Marine Aggregates Industry

Marine conservation is very much in its infancy although both statutory and voluntary Marine Nature Reserves have been operating since the 1970s. The UK Marine SACs project was formed in 1996 to promote the implementation of the Habitats Directive in marine areas through setting up management schemes at 12 candidate marine sites. A ruling given by a UK High Court Judge in November 1999 that the Habitats Directive applies beyond the 12nm territorial limit up to 200 miles from the coast, has given added impetus to the setting up of marine SACs.

As is apparent in the review of remediation methods in Section 5, habitat restoration and enhancement within the marine environment is in an experimental stage. The concept and indeed the place for remediation and habitat creation in the marine environment are under discussion within the wider context of marine conservation policy development.

Conservation objectives for the marine aggregate industry in recent years have focused on minimising the impact of dredging thereby improving / accelerating natural recovery processes.
The potential impacts of marine aggregate dredging on nature conservation priorities and biodiversity form an integral component of the environmental impact assessment (EIA) which is currently required under the Government View Procedure (see 2.5.2 below) for all new licence applications. Unavoidable impacts are minimised through appropriate mitigation, management and monitoring.

The importance of this precautionary approach in a debate on remediation should not be overlooked. Applying the adage, prevention is better than a cure, it can be argued that the prevention of unacceptable impacts in the natural environment has greater merit than reaching a point where intervention in the natural recovery process through remediation becomes necessary. This still begs the question: what is unacceptable in terms of impact? This issue will be considered further in section 6.

The marine aggregates industry contributes to the supply of environmental data for marine and coastal areas. As a greater understanding of anthropogenic impacts, marine organisms and habitats is obtained, this will have an input into biodiversity strategies within the marine environment.

The activities of the marine aggregate industry are considered in several of the Maritime Habitats Action Plans (HAPs) and species action plans (SAPs) within the UK BAP specifically:

- Sheltered muddy gravels HAP;
- Sublittoral sands and gravels HAP;
- *Sabellaria spinulosa* reefs HAP;
- *Atrina fragilis* SAP.

Inevitably a management plan which seeks to:

"*protect the extent and quality of a representative range of sublittoral sand and gravel habitats and communities*"

will need to consider closely the impact of the marine aggregates industry on these habitats and the measures that may be necessary to preserve their biodiversity. The importance of *Sabellaria spinulosa* reefs has already been recognised in licence conditions attached to aggregate extraction areas and the SAP reinforces the protection this species should be given. The rarity of *Atrina fragilis,* one of the largest European bivalve molluscs, may be a lesser known concern. It is speculated that aggregate extraction historically may have been a contributory factor in the reported decline of this species which occurs in mud, sand and gravels from below MLWS to depths of 400m.

By reviewing existing knowledge of the impacts of the marine aggregates industry on a range of habitats, this study will offer a source of reference to regulatory authorities, industry and other agencies on the measures necessary to manage potential biodiversity impacts including the appropriateness of remediation.

2.5.2 Conservation Legislation & Policy affecting the Marine Aggregates Industry.

The European Community Habitats Directive and Birds Directive / The Conservation (Natural Habitats &c.) Regulations1994. The marine aggregates industry is subject to the Conservation (Natural Habitats etc.) Regulations 1994, in relation to the degree to which dredging activities may affect coastal, marine and future offshore Natura 2000 sites (SPAs, SACs and Ramsar sites).

The industry has to consider the potential impacts to habitats defined under Annex I of the Habitats Directive (of which four, sandbanks which are slightly covered by sea water all the time, reefs, submarine structures made by leaking gases, submerged or partially submerged sea caves, are known to, or potentially, occur in UK offshore waters); protected species defined under Annex II of the Habitats Directive; and migratory and Annex I bird species defined under the Birds Directive. Offshore designations of these habitats and species are currently being considered by the Joint Nature Conservation Council (JNCC) and have the potential to impact upon the activities of the marine aggregates industry (Johnson & Tasker 2002). Activities within 12nm of the coast are subject to the same directives as interpreted by guidelines laid down by English Nature (UK Marine SACs Project 2001).

Applications for production licenses have been considered under a **Government View (GV) Procedure** since 1968. The current Interim Government View Procedures in England are controlled by the Minerals and Land Reclamation Division of the Office of the Deputy Prime Minister (ODPM, formerly DTLR). The interim GV procedure is shortly to be replaced by a statutory system, which will transpose the provisions of the Environmental Impact Assessment and Habitats Directives, insofar as they relate to marine minerals dredging, into UK law.

These new **Environmental Impact Assessment and Habitats (Extraction of Minerals by Marine Dredging) Regulations** will apply to England, Wales and Northern Ireland.

The current GV procedure requires the undertaking of an Environmental Impact Assessment (EIA) and associated scientific studies including a Coastal Impact Study (CIS) and benthic survey for each application. This EIA process is supported by an extensive process of consultation with government departments, statutory advisors and wider stakeholders, relevant government bodies and the general public. Provision is also made for a public inquiry, if necessary.

A favourable GV is needed before a production licence is issued by the Crown Estate. An application for a GV must be made to the Office of the Deputy Prime Minister (ODPM) or the Welsh Assembly Government (WAG) as appropriate. In reaching its decision ODPM or WAG will consider all information submitted with the application, including reports on environmental effects of the proposed dredge and all comments received following consultation and advertisement of the application.

Figure 2.6: Marine Aggregate Dredger
Source: BMAPA

A favourable GV is unlikely if an application could have a significant adverse effect on benthic ecology, fisheries, archaeological or nature conservation interests, physical processes or have an impact at adjacent coastlines. Consequently the GV procedure acts as a filter in the licensing process. Proposed extraction which would have a significant adverse impact is unlikely to achieve a favourable GV and be granted a licence. The GV may also suggest licence conditions to mitigate impacts as a result of extraction, including the zoning of licence areas, and monitoring of the area, pre, during and post dredging.

Current government policy on the extraction of marine sand and gravel from the seabed are contained in Marine Minerals Guidance Note 1.

Marine Minerals Guidance 1 (MMG1) *'Extraction by dredging from the English seabed'*, reaffirms the government wish to see the continued use of marine dredged sand and gravel, to the extent that this remains consistent with the principles of sustainable development. It acknowledges the need for investment in the industry and continued access to resources, while stressing the importance of avoiding significant harm to the environment, fisheries or unacceptable effects on other sea users. The Government believes this can be achieved by appropriate mitigation including:

- Minimising the total area licensed / permitted for dredging;

- The careful location of new dredging areas □ embodying a precautionary approach, close scrutiny is given to areas which might have an adverse impact on:
    i. fish spawning, migration routes or as nursery and over-wintering periods;
    ii. designated conservation sites;
    iii. war graves, wrecks or sites of archaeological interest;
- Considering all new applications in relation to the findings of an Environmental Impact Assessment (EIA) where such an assessment is required; currently the view is that this will be required for all new dredging applications;
- Adopting dredging practices that minimise the impact of dredging; including zoning which further subdivides areas to restrict dredging activity to discrete zones;
- Requiring operators to monitor, as appropriate, the environmental impacts of their activities during and on cessation of dredging and;
- Controlling dredging operations through the use of conditions attached to the dredging licence or dredging permission.

The policy stipulates a number of specific issues which require consideration within the EIA, these include:

- The need for seasonal or tidal restrictions to protect sensitive features,
- The appropriateness of on-board screening;
- The removal of overburden (fine material which overlays the aggregate resource);
- The potential cumulative effects of interaction with impacts from other sites or other activities such as fishing, pipeline discharges or disposal of harbour dredgings.

Stringent conditions may be applied under the GV such as governing the volumes dredged, or the times, locations or methods by which material is extracted, in order to minimise and manage potential environmental impacts.

3.0 The Impacts of Dredging on Habitats and Recovery of Benthic Communities

3.1 Introduction

Section 3.2 identifies and defines the habitats and biotopes found in association with marine aggregate sites. The principal methods of dredging are identified in Section 3.3, and the spatial extent, intensity and frequency of dredging activity defined. The impacts of aggregate dredging upon benthic communities for the three habitats types considered as typical of current UK dredging activity are discussed in Section 3.4, in terms of fauna type, dredging technique used, dredging intensity and frequencies and using information obtained through literature review and expert judgement. The recovery of the various habitats and faunal types, post-dredging, are discussed in section 3.5, with reference to the various scale of impacts identified previously.

3.2 Habitats and Biotopes

Three major habitat types have been identified that are considered to best describe the broad range of conditions and biotopes associated with marine aggregate extraction sites around the UK. These are characterised as follows:

- Shallow water mobile sands;
- Shallow water stable gravel with transient sand;
- Shallow/Deep water stable gravel.

The nature of each of these habitats is briefly described in the following sections, based on a review of the available literature on UK and near-continental dredging sites.

3.2.1 Shallow Water Mobile Sands

These comprise classical mobile sand banks or large mobile, sandy bedforms with both localised/superficial sediment mobility and in some cases longer term movement of the sand bank features. Examples of these include sites in the outer Bristol Channel (e.g. Nash Bank) and the East Coast (e.g. Cross Sands and the sand banks systems off Lowestoft & Great Yarmouth).

Examples of JNCC defined biotopes that may describe these habitats include the largely afaunal biotope, Infralittoral Gravel and Sand.MobRS (**Sparse fauna in reduced salinity infralittoral mobile sand**) and the more diverse Infralittoral Gravelly Sand.Ncir (***Nepthys cirrosa* and fluctuating salinity-tolerant fauna in reduced salinity infralittoral mobile sand**). However, since marine aggregate dredging is typically recorded from coastal areas of full salinity, these biotope descriptions may not fully apply.

Biotopes in similar habitats which have been defined by Brown, et.al. (2001), include **Clean mobile sand with *Abra prismatica*** and **Sand and gravelly sand with *Ophelia borealis, Bathyporeia* sp. and *Pomatoschistus minutus*** both found at Shoreham on the UK south coast.

Species typically recorded from mobile sandy habitats as a result of marine aggregate dredging EIA or monitoring programs include the mysid *Gastrosaccus spinifer*, amphipods such as *Pontocrates arenarius*, polychaetes such as *Ophelia spp*. and *Nephtys spp*. and sandeels

The essential feature of these habitats is the high degree of sediment mobility, which in its most extreme form leads to afaunal sediments and fauna adapted to natural, physical disturbance. Typically species diversity, abundance and biomass are naturally very low.

3.2.2 Shallow Water Stable Gravel with Transient Sand

These include stable gravel features that are subject to the regular, transient movement of sand as sand ribbons/waves, etc. Good examples of these are found in the southern North Sea e.g. sites off the Suffolk/Norfolk coastline (e.g. the Great Yarmouth block of licensed dredging areas) and off the Klaverbank, in slightly deeper water.

Biotopes related to these habitats include the JNCC defined, Circalittoral Mixed Sediments. ModMx, (***Modiolus modiolus*** **beds on circalittoral mixed sediments)** and Circalittoral Mixed Sediments. SspiMx (***Sabellaria spinulosa*** **and** ***Polydora*** **spp. on stable circalittoral mixed sediment)** (please note that this does not include *Sabellaria* reefs)**.**

Brown et.al. report a similar biotope to the former, **Mussel beds on mixed, heterogeneous sediments** as well as **Echinoderm dominated *(Echinocyamus pusillus*** **and** ***Psammechinus miliaris*****) gravelly sand with occasional sand veneer.** Several of these biotopes are tolerant or indeed require the presence of mobile sands to develop, e.g. the S*abellaria* based biotopes.

Typical species recorded from these sand-affected gravelly habitats during aggregate dredging EIA or monitoring studies include a mixture of species characteristic of both mobile sandy sediments, such as the polychaetes *Ophelia limacina* or *O. borealis, Nephtys spp*., string swimming amphipods such as *P. arenarius, Haustorius arenarius* and *Bathyporeia spp* as mentioned previously together with species indicative of coarser sediments such as the bryozoans *Conopeum reticulum*, and *Electra pilosa* and hydroid species such as *Sertularia cupressina*. Of particular note in such areas are species which are scour-tolerant or indicative of scour such as horn-wrack *Flustra foliacea,* sea chervil *Alcyonidium diaphanum* and the Ross worm, *Sabellaria spinulosa*.

The essential feature of these habitats is a community generally characterised by robust mobile opportunistic species adapted to, or tolerant of, disturbance through abrasion and temporary deposition of transient sediment which have a high rate of recolonisation and growth. Such areas are seldom found to support slow-growing, long-lived 'equilibrium' species (with the exception of *Flustra* and *Sabellaria*) and typically lack a significant epifaunal community. Such habitats are found widely distributed around the UK in relatively shallow water areas subject to regular wave and/or strong tidal currents

3.2.3 Shallow/Deep Water Stable Gravel

It is considered that two types of 'stable' gravel exist, shallow and deep water, although data on the latter is limited and impacts of dredging in both are similar. For this reason they will be treated as one type for the purposes of this review.

This habitat in shallow conditions includes stable seabed features with little localised or superficial mobility, e.g. the Hastings Shingle Bank and various sites around the Isle of Wight and along the south coast in general.

Biotopes that may be found in shallow habitats include the JNCC defined Moderately Exposed Circalittoral Rock. Flu.SerHyd, (***Sertularia argentea*****,** ***Hydrallmania falcata*** **and** ***Flustra foliacea*** **on tide swept mixed substrata)**. Brown, et. al. defined biotopes SH and HB, **Cobbles with algae (unidentified) and** ***Crepidula fornicata*** and **Coarse gravel with attached epifauna** respectively.

The deeper water version of stable seabed includes the currently proposed dredging sites from the Eastern English Channel. The JNCC are currently redefining the biotopes pertinent to this type of habitat so no completed examples currently exist. However, many of the existing Moderately Exposed Circalittoral Rock biotopes may well be included, such as those found in the shallow gravel habitats, for example the general group MCR.ByH, (**Bryozoan/Hydroid turfs (sand influenced)**) and the more specific MCR.Oph, (***Ophiothrix fragilis* and or *Ophiocomina nigra* beds on slightly tide-swept circalittoral rock or mixed substrata).** Although sand and silty sediments may influence these biotopes, they are essentially stable particulate habitats, with little or no superficial fine sediment movement.

Typical species recorded from these deeper water, relatively stable gravel habitats during aggregate dredging EIA or monitoring studies include *K* strategist 'equilibrium' species, which are long lived and slow growing but competitively superior species and which invest in few well developed juveniles. Opportunist or *r*-strategist species are often not present within this type of community since they would likely be excluded by the competitively superior K-strategists. Common fauna within these areas include large mobile animals and deposit and suspension feeders such as the razor shell, *Ensis sp*., scallops, *P. maximus*, dog cockle, *Glycymeris glycymeris*, sea urchins, *Echinocardium sp.* and *Echinus sp.*, edible crab, *Cancer pagurus*, and larger burrowing crustaceans such as *Upogebia* sp. These areas are also able to support diverse sessile epifaunal communities characterised by species such as dead men's fingers, *Alcyonium digitatum* together with a range of anemones and erect hydroids and bryozoa.

The essential feature of these stable gravel habitats is the establishment of a diverse, species and biomass-rich community which has been allowed to develop under stable environmental conditions where the faunal community structure is controlled by biotic factors, such as competition and predation, rather than by abiotic factors, such as environmental stress through regular natural disturbance resulting from, for example, sediment scour. The relatively low natural disturbance is often associated with deeper, more quiescent conditions than the sand-scoured gravel areas characteristic of more shallow inshore areas, though the seabed substratum is often similar in composition.

3.2.4 Other Dredged Habitats

Other habitats exist that may have been dredged in the past, but are currently not considered to be relevant to this study. These habitats are no longer subject to dredging in the UK due to environmental constraints and include:

- **Shallow water mobile gravel** □ Generally very shallow water features with localised/superficial mobility, i.e. wave exposed conditions.

- **Shallow water stable/sheltered sand** □ Static sand bank features but with localised/superficial mobility, generally estuarine, sheltered coastal environments, e.g. the Winner Bank off Langstone Harbour mouth in the Solent.

3.3 Dredging Activities

Two principal methods of dredging are employed in marine aggregate extraction: **static dredging** and **trailer dredging**. It should be noted that these two methods may have potentially different impacts on the seabed environment.

The current UK marine aggregate dredging fleet is dominated by **trailer suction hopper dredgers**, which employ a single, rear-facing pipe with a centrifugal pump mounted either within the hull of the vessel, or on the dredge pipe itself. Depending upon the size of the dredger, between 1200 and 8500 tonnes can be loaded in a single cargo.

As the name suggests, the trailer dredging process involves the dredge pipe being lowered to the surface of the seabed until the drag head rests on the seafloor. As the vessel moves slowly forwards (around 2 knots) trailing the draghead, seabed sediments and water are lifted from the seafloor by the pump, and pass up the dredge pipe into the hold of the vessel. The sediment is retained, while the excess water generated by the dredging process is drained from the hold and returned to sea via spillways.

As the drag head passes over the seabed, a cut of sediment around 3 metres wide and 15-30cm deep will be removed. As this technique is suited to more extensive sheet-type deposits of sand and gravel, vessels may have to make repeated passes over the same area in order to load a full cargo. The inter-crossing trailer tracks create an irregular seabed topography, and over time, the dredging process will gradually lower the surface of the seabed across the licence area.

While the principal method of dredging is trailing, where an aggregate resource is spatially restricted or locally thicker, a number of the vessels in the UK fleet are also able to **static dredge** - anchoring, or remaining stationary over the deposit, while loading. Over time, static dredging creates a more localised, saucer shaped depression, typically 8-10m deep and 50m to 200m in diameter. The modern trend is for multi-purpose trailer suction hopper dredgers that are also able to static dredge. There are limitations though, and not all trailer dredgers are able to load in this manner because of vessel size, handling characteristics and gear type (drag-head type and pump location).

The construction industry typically requires marine aggregate to be supplied with a gravel content of greater than 50%. Where the in-situ gravel content of the resource being dredged falls below this, vessels can employ on-board screening to alter the gravel content of cargoes.

Static screen boxes or screening towers are used to alter the composition of the dredged aggregate retained, by passing the water/aggregate mix over a mesh screen. Assuming that the intention is to increase the gravel content, a proportion of the finer aggregate and water will pass through the screen, and be returned to the sea by means of a reject chute. The remaining fine aggregate, together with the coarse element and remaining water enter the vessels cargo hold. This process can be reversed, if the intention is to produce a sand cargo, with the coarse element of the dredged aggregate being rejected.

The dredging technique to be employed and the ability to screen will be determined within the Government View permission granted for each licence area, dependant upon site-specific sensitivities.

The two types of dredging generally represent high spatial intensity and lower spatial intensity respectively. In addition to the major types of dredging techniques, several different sub-types may be defined based on other characteristics of the dredging activity.

Spatial extent and **intensity** - ranging from small, highly localised areas of intense activity (generally experimental dredge areas, or discrete areas in larger extraction blocks) to low/high intensity activity spread over a larger area, such as those typically found in many aggregate extraction areas. Figure 3.1 shows the results of a bathymetric survey for a former static dredging site on the Inner Owers bank. The mottled appearance is formed from static dredging depressions which are typically about 6 m deep and up to 50m across.

Frequency of activity – These may be defined on the basis of the number of visits per year (which broadly correspond to the total tonnage extracted per year) or period since the last visit. As no specific data are available for many of the studies, three general levels of activity have been employed to assess the data; low, medium and high. These correspond to:

- **High** frequently to occasionally within the same year;
- **Medium**- sporadically within one year or with more than a one year break but less than a two year period since last dredged;
- **Low** - more than two years since last dredged.

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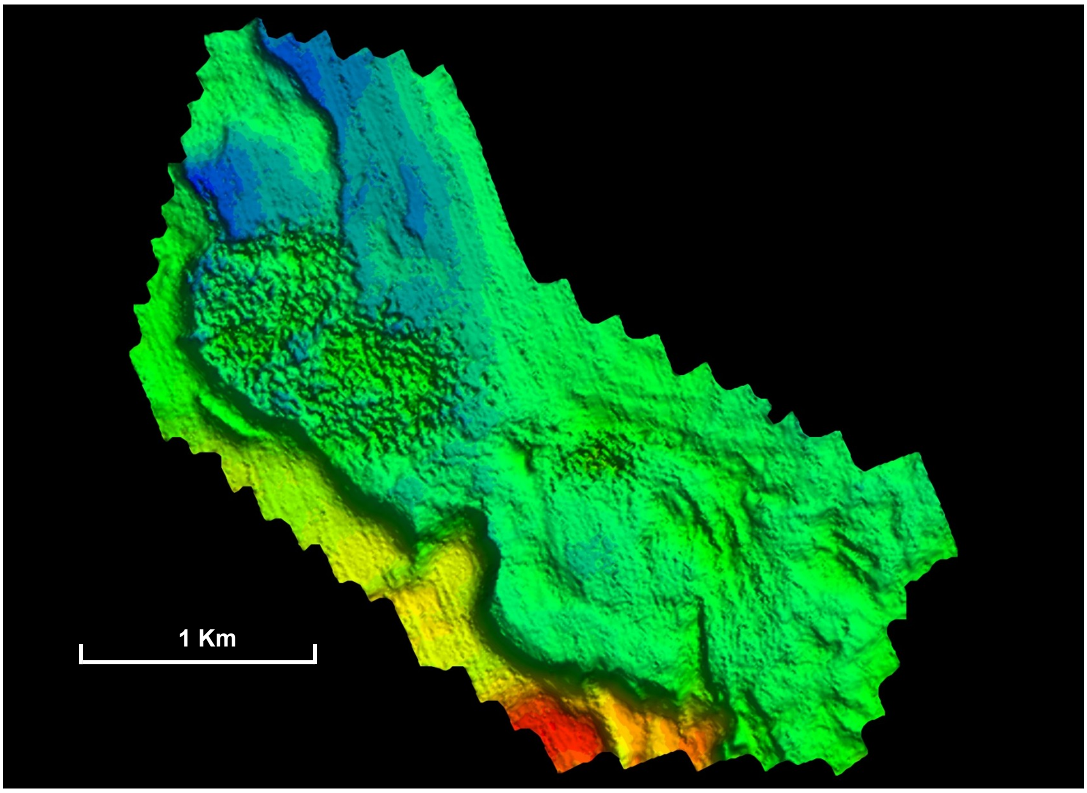


Figure 3.1 Swath bathymetry image showing former dredged area of Inner Owers Bank. The heavily mottled area is evidence of former static dredging activity. Source: Emu Ltd.

In defining the levels of frequency of dredging described above, it should be noted that this crude categorisation has been driven by the lack of specificity in the available data from the scientific literature examining impacts of dredging on the benthos and the subsequent recovery process at UK and near continental sites. These frequencies may not adequately describe the realistic dredging intensities which actually occur in some habitats or at some licensed areas and therefore may not describe in a representative manner dredging impacts on benthos and recovery in UK waters.

3.4 Impacts of Dredging upon the Benthos

The impacts of dredging upon the principal habitat types are summarised below. Details on the bibliographical sources of these findings are contained in two tables in Annexes A and B. The **Impacts Table** (Annex A) gives observed primary and secondary impacts in terms of species composition, broken down into the main habitat types based upon dredging intensity / frequency.

The impacts considered include not only the direct effects but also **peripheral consequences** of the dredging activity e.g. screened sand sediments and bedload impacts outside the licence area. The **Recoverability Table** (Annex B) provides a breakdown of recovery for the main habitats based upon the history or dredging intensity/frequency or type of dredging. Gaps in the information, or areas of uncertainty, have been identified.

3.4.1 Impacts in Shallow Water Mobile Sands

Dredging at sand extraction sites results in measurable impacts on bottom fauna, including reductions in species diversity and abundance. Static dredging methods appear to have the most notable impact on bottom fauna at sand sites, whilst trailer dredging impacts may include a 50% reduction in species diversity and 60% decrease in species abundance (Newell *et al,* 1998). Greater losses in species diversity and abundance may occur at more intensively trailer dredged locations (Poiner & Kennedy, 1984 & Sardá *et al*, 2000).

Low frequency, low intensity and spatially widespread dredging activities generally allows for some recolonisation between dredging periods by the typically mobile sand species, and may retain pockets of localised fauna between dredge tracks or depressions for larval recruitment and adult immigration (Newell et al 1998; Van Moorsel 1993,1994)

Figure 3.2: *Pomatoschistus minutus* (Sandy Goby)
Source: British Marine Life Study Society

A reduced dredging effort over a wider area tends, therefore, to result in a comparatively limited impact since the fauna around the dredged area would contribute to the post dredge diversity, abundance and biomass, which are typically low and spatially variable in these kinds of mobile sandy habitats.

In contrast, high frequency, high intensity and spatially concentrated dredging activity would tend to prolong the disturbance and eliminate the majority of the fauna within the dredge site, as a result of direct removal by the drag head (Boyd & Rees 2003b; Boyd et al 2003a) Nonetheless, the highly mobile fauna typical of this type of habitat would still be expected to provide a source of recruits to the dredged area from surrounding, unaffected areas (Newell et al 1998).

The review of the literature for these mobile sandy habitats highlights the key characteristics of the biotope and the typical species recorded. These are predominantly mobile, opportunist infaunal/epifaunal species, such as polychaete worms and amphipod crustaceans, which are highly fecund, capable of high rates of recruitment and recolonisation and resilient in response to the high level of natural disturbance experienced in these habitats. This adaptation is reflected in the flexible body forms, ability to maintain position in relation to the sediment surface and their rapid burrowing capability (Posford Duyvier Environment & Hill M.I. 2001)

This natural ability to constantly adapt to a highly unstable, naturally disturbed habitat means that the impacts of dredging on these mobile sandy areas is often short-lived and temporary with rapid recruitment by both adults and larvae, provided that the underlying nature of the sand habitat (in terms of particle size for example) is not fundamentally changed by the dredging operations. However, the mobility of these habitats means that any dredge tracks or depressions will tend to be rapidly infilled by sandy material of the same nature as that removed by dredging so that the physical effects on the seabed are relatively short-lived.

Dredging may also cause more indirect effects through the creation of fine sediment plumes or bedload sediment transport resulting from overspill or screening. Such peripheral impacts have also been described in the literature with increased faunal diversity and abundance up to 2km from a dredged sand site (Poiner & Kennedy. 1984) and a significant enhancement of biomass (Newell *et al* (2002) reported. These increases may be due to the release of organic matter by dredging activity and the transportation of resources by the dredge plume.

In general, however, the mobility of the natural habitat (and the associated natural tolerance of the fauna) and the generally very low fines content of many sand bank aggregate resources means that impacts resulting from these secondary aspects would not normally be highly significant.

Summary of Impacts in Mobile Sand Biotopes

- Mobile sand is characterised by mobile opportunist infaunal species, e.g. polychaete worms and amphipod crustaceans, capable of high rates of recruitment and recolonisation. The fauna is also naturally tolerant of sediment disturbance and are robust to sediment instability due to dredging and smothering in adjacent areas.
- Direct impacts of dredging may result in a reduction in species diversity and abundance. Static dredging has the most significant impact. Impacts are comparatively short-term with rapid recolonisation by naturally mobile/opportunistic fauna.
- Trailer dredging typically leads to a 50% reduction in species diversity, and 60% decrease in species abundance. These values can increase at sites where high intensity, high frequency and concentrated dredging occurs.
- Spatially widespread dredging can retain pockets of benthos between dredge tracks for larval recruitment and adult migration. Reduced dredging effort will result in comparatively limited impact as these survivors will contribute to post dredging diversity, abundance and biomass measures.
- Rapid in-filling of dredge tracks/depressions (resulting from either slumping of the sides or deposition of particulates) with little long term change to the habitat generally recorded.
- Peripheral impacts may occur but are not normally considered significant due to natural mobility and sand scour of the habitat and low fines content.

3.4.2 Impacts in Shallow Water Stable Gravels with Transient Sand

The literature reveals that the significance of dredging impacts, within this general habitat type, varies according to the fauna naturally inhabiting the area, the nature of the seabed including both particulate size ranges and stability and the intensity and frequency of the dredging.

Primary impacts on the benthos reported by the literature include reductions in species number, abundance and biomass. Values cited range from 30-80% reduction in species; 70-90% reduction in abundance and 80-90% reduction in biomass in the area dredged compared with either baseline or control site values. These levels of impact are generally related to high frequency and/or high intensity dredging with reductions in impact likely where dredging activities are reduced either in the periods between dredging occasions (where partial recovery can occur) or as a result of the area being subject to the extraction of lower tonnages, leaving more of the seabed in the area unaffected by direct drag head disturbance. The species most seriously affected and which are generally slower to recover include sessile epifauna unable to avoid the disturbance or intolerant of smothering and slower growing equilibrium species such as large bivalve molluscs.

However, in this habitat type **gravel subject to disturbance by naturally occurring mobile sediments**, such impacts on the benthos tend to be relatively short term since the majority of fauna are mobile and/or tolerant of temporary smothering and raised levels of abrasion by mobile sediments which typify the natural environment.

In addition, the habitat has been noted in the literature as being typified by low diversity and often comprising an opportunistic fauna dominated by polychaetes and crustaceans, a reflection of the harsh natural environment which tends to exclude the establishment of more diverse communities and specifically the diversity of epifaunal and larger, slow growing species (e.g. Newell *et al*, 2002).

Figure 3.3: *Flustra foliacea* (Hornwrack)
Source: Encyclopedia of Marine Life of Britain and Ireland

The naturally suppressed diversity of these habitats and the adaptation of the fauna to natural disturbance and inundation by mobile sands means that recovery post-dredging will also be relatively rapid compared to more stable gravels, particularly in terms of diversity and abundance. Biomass, which becomes suppressed in area subject to dredging is reported to require a longer timespan for recovery compared with species diversity and abundance (van Moorsel and Waardenburg, 1991; Desprez, 2000; Newell *et al*, 2002).

Dredge tracks or depressions in the underlying gravel can persist for some time but will ultimately tend to be infilled by the mobile sand moving over these habitats. This change from gravelly seabed to a finer, sandier seabed will tend to result in the more sandy fauna dominating in these areas and a corresponding loss of the fauna more typical of the gravelly areas.

With regard to peripheral impacts, recent studies have reported areas of enhanced biomass up to 2km from the dredged area along the tidal axis resulting from enrichment by organic matter released from screening and/or the benthic plume from the dredging activity. Impacts from coarser sediment, such as sand, released by screening in this habitat are not likely to be significant away from the immediate dredged areas because of the natural adaptation of the fauna to inundation and scour by mobile sands.

An additional issue only occasionally reported in the literature (and presumably therefore not typical of most dredged sites) was the development of anoxic conditions resulting from an accumulation of finer sediments particularly in depressions created by static dredging. This effect, where it occurs, would tend to reduce species diversity.

Summary of Impacts in Shallow Water Stable Gravels with Transient Sand

- Primary impacts on benthos include reductions in species number, abundance and biomass. Levels of impacts generally related to frequency and intensity of dredging.
- For this habitat, the suppression of benthos is generally short-term with the majority of fauna naturally tolerant of temporary smothering and raised levels of abrasion by mobile sediments.
- Dredge tracks can in-fill within months to several years. This can result in areas of seabed dominated by finer sediment, relative to surrounding areas, with an associated shift in the distribution of faunal community in the habitat complex.
- Temporal enhanced biomass up to 2km along tidal residual can occur, as a result of enrichment by organic matter released from the dredge plume.

3.4.3 Impacts in Stable Gravel Habitats

The principal impacts of dredging in more stable gravel habitats have been clearly demonstrated at several sites where dredged areas have become almost totally devoid of fauna either where static dredging occurs (Shelton and Rolfe, 1972), or where repeated and very intensive trailer dredging has occurred (Emu Ltd. 2002).

In general, static dredging has a more pronounced impact but over a very small area, whilst trailer dredging results in a reduction in diversity of circa 70%, with both biomass and abundance reducing by up to 90%. Continuous high frequency or large volume dredging results in the persistence of these levels of impact. Where reduced frequency or spatially less intense dredging occurs, then impacts are reduced, either through partial recovery between dredging occasions or because of a lower, initial dredging intensity.

The species groups which tend to be most vulnerable to significant damage by dredging are the slow-growing, delicate, sessile epifauna unable to tolerate disturbance or smothering, for example anemones, bryozoans and hydrozoans in general. However, in these stable gravel habitats, intensive dredging activity can also disturb the more complex ecosystem relationships that characterise these stable ‘climax’ gravel habitats. Examples of this complexity might include the presence of large, slow-growing species (e.g. scallops) which subsequently create micro-habitats or niches for a variety of other invertebrate infaunal and epifaunal species.

The role of dredging intensity in the significance and ultimately the recoverability of impacts on these stable gravel habitats is highlighted by a number of recent studies. High intensity trailer dredging or static dredging will remove almost all of the fauna in an area over time, whereas lower intensity dredging will tend to leave relatively unaffected pockets of the stable gravel communities between dredged areas.

Over time, both static and trailer dredging can lead to a range of impacts on the seabed in these otherwise stable gravel areas, including the following:

- Fine sediment deposition on the otherwise un-impacted edges as a result of screening and overspill;
- Destabilised sediments on the steep sided trench wall and;
- Faunally devoid mobile sediments in the bottom of the dredge tracks or pits (again this mobile, fine sediment is generally derived from screening or overspill).

The impacts on the seabed and associated faunal communities viewed from this perspective can be substantial where they occur, but are often relatively spatially discrete. However, long-term, intensive dredging means that significant impacts can occur over wider areas that may result in longer term changes to these habitats.

Figure 3.4: *Psammechinus miliaris* (Green Sea Urchin)
Image: Sue Daly (Published on the *MarLIN* Website)

Such changes to the habitat, listed above, have considerable significance when considering remediation and recovery, particularly in the deeper water stable gravel environments. There is currently no data available that has monitored the effects of dredging on the complex stable gravel communities which characterise sites at water depths of 30m or more such as the Eastern English Channel. A theoretical assessment of the impact of dredging in such areas has been attempted as part of the EIA process based upon the existing knowledge of the physical and biological processes which occur in shallower water environments and what is known of the benthic communities and environmental conditions at these greater depths. Newell *et al* (2002) suggest that the limitation of sediment disturbance by wave action and storm events mainly to depths of less than 30m will be likely to cause the persistence of dredged features such as trailer dredging tracks or static dredging depressions. They describe deep water gravels as supporting "equilibrium" communities with complex species interactions, similar in nature to some of the shallower water stable gravel areas but with even greater community complexity. The presence of a comparatively higher number of longer lived and slower growing species is likely to significantly extend the recovery of the communities, even compared to the shallower gravel habitats. For example, Dog cockles, *Glycmeris glycmeris* in the Eastern English Channel may be up to 14 years old with recruitment by juveniles at approximately 5 year intervals. Recovery of a Dog Cockle population following dredging may take as long as 15-20 years.

A key issue for these stable gravel habitats, particularly in deeper water areas, is the secondary impact of dredging and specifically the sandy sediment rejected by screening and overspill processes settling to the seabed. Studies suggest that much of this sandy sediment remains within the licensed area where it will tend, over time, to be transported away along the tidal axis to adjacent areas of seabed as small bedforms. This process will tend to result in a gradient of effects on the gravel communities outside of the immediately dredged areas ranging from almost continual smothering close to the dredging activity to intermediate smothering or scour as the sand is transported further afield. Recent studies suggest measurable effects resulting from this bedload material up to 2km from the dredging operations, although the area affected and the significance of effects will be related to dredging intensity and screening rates, frequency, nature of the prevailing currents and the resource material being dredged. Ultimately, such effects could result in the development of a habitat similar in nature to that previously considered shallow water gravels affected by transient sands in place of the stable, climax gravel communities that previously existed, with the corresponding decrease in diversity, complexity and productivity.

These secondary impacts can be of significance, with regard to the current study areas for two principle reasons:

- An extension of the impact area beyond the area of actual dredging activity at the seabed;
- A potential extension of normal recovery periods because of changes to the benthic habitats.

Specifically, it is possible that, over time, intense dredging activity could result in a significant area being affected by these secondary impacts which substantially enlarges the area of seabed affected by the dredging operations. In addition, the recovery process of the stable gravel habitat would be extended because of the change in the nature of the seabed from a stable gravel to a mobile sand. In order for full recovery to the pre-dredge baseline, these changes to the physical environment would need to be substantially reversed through natural processes.

Summary of Impacts on Stable Gravel Habitats

- Where static or repeated intensive trailer dredging occurs, dredged areas can become almost totally devoid of fauna, with static dredging having the most pronounced impact although spatially limited.
- Trailer dredging can reduce species diversity by ~70% and biomass by ~90%. Reduced frequency or spatially less intense dredging decreases impacts, either through recovery or reduced initial damage.
- Species groups most seriously damaged are sessile epifauna unable to tolerate disturbance and/or smothering, e.g. anemones, bryozoans, hydrozoans. Complex community structures can also be disturbed.
- Where trail dredging has left deep trenches or static dredging occurs, additional impacts arise including; limited sediment deposition on un-impacted edges; destabilised sediments on steep sided trench walls; and mobile sediments devoid of fauna in the bottom of the tracks or depressions.
- The oretical dredging impacts to benthos on deep water (>30m) complex stable gravel, suggests that the presence of higher numbers of longer-lived and slower-growing species is likely to significantly extend the recovery time compared to shallower sites. For example Dog Cockles in the Eastern English Channel may live for 14 years with recruitment of juveniles at 5 yearly intervals; as a result the recovery of a population may take 15-20 years.
- Secondary impacts from sandy sediment generated by screening or overspill can extend the areas of impact and the timescales for recovery, related to changes to habitat, sediment type and sediment stability.
- The degree of primary and secondary impacts seems to be closely linked to dredging intensity and screening rates, frequency of dredging, nature of the prevailing tidal currents and the resource material being dredged.

SUMMARY OF DREDGING IMPACTS ON BENTHOS

- Dredging generally leads to a reduction in species diversity, abundance and biomass.
- Static dredging has the most adverse impact and can leave the bottom of depressions devoid of fauna, but occurs over spatially limited areas. Trailer dredging can have a less intensive effect but over a wider area.
- The extent and significance of the impacts on all benthic habitats is related to dredging intensity, frequency and the nature of the existing environment, but with the effects greatest in more stable, deeper water gravel habitats.
- Destabilisation of stable gravel habitats can lead to increased mobile fine sediment and can potentially alter seabed sediment composition and stability with an associated shift in faunal community. This is also most significant for deeper water, stable gravel communities.
- Sessile epifauna e.g. bryozoans and hydrozoans, and slow growing species e.g. large bivalve molluscs, tend to be amongst the species most significantly affected by both primary and secondary effects of dredging. These species are most common and have greatest community significance in the deeper water stable gravels.
- The significance and recoverability of impacts on the benthos is closely related to the stability of the natural environment within which dredging occurs and therefore the degree of natural adaptation of the fauna to physical disturbance.
- Temporary enhanced biomass can occur up to 2km from the dredge site along the tidal axis due to the release of organic matter by dredging activity.

3.5 Recovery of Benthic Habitats

The process of recovery following environmental disturbance is generally defined as ***the establishment of a community that is similar in species composition, population density and biomass to that previously present or at non-impacted sites.*** (C-CORE 1996 as cited by Newell 2002). A range of factors affect the rate of recovery of a faunal community following disturbance. However, in general, sediment type and stability will tend to be amongst the most important, as these variables tend to reflect the prevailing environmental conditions. Sediment characteristics also largely determine the composition of the infauna / epifauna, as well as being most likely to be affected by dredging operations. Recovery has therefore been considered in relation to the major habitat types described at the beginning of this chapter.

3.5.1 Recovery of the Benthic Fauna in Shallow Water Mobile Sands

Recovery of bottom fauna in shallow water mobile sands is generally rapid and follows a predictable pattern of mass import of opportunist species, recolonisation and community stabilisation to pre-dredge conditions, once dredging ceases. If the sediment composition and structure post-dredging does not significantly alter from its pre-dredged state, it is likely that recovery and recolonisation of the same benthic faunal communities will occur (Van der Veer (1985), De Groot (1986)).

The rate at which the biological recovery occurs is dependant on the hydrodynamic regime of the local environment, the duration and depth of dredging, and the rate of species recruitment. In dynamic environments which tend to typify UK sand dredging areas, recovery of the bottom fauna commences within a few months or even weeks with full recovery of the benthos highly likely within 1 - 4 years, depending on the extent of the impact. Studies at the Nash Bank off the coast of South Wales have indicated no statistical difference between mobile sand fauna within dredged and undredged parts of the bank, even whilst dredging is ongoing, reflecting both the scarcity of the fauna and the high mobility and rapid recolonisation in these habitats (Emu Ltd, 2002d).

Recovery will start after the cessation of dredging, but may even include some recolonisation between dredging loads as apparently evidenced by the Nash Bank study. Recolonisation of dredged sands may be immediate, with some robust fauna settling from the dredger overspill and screening material e.g. bivalves. Adult migration and passive larval import is also important to the recolonisation processes, the rates of which are governed by the magnitude of the impact. Recovery after small operations, for example, has been recorded to occur within 28 days (de Groot, 1986) and Lopez-Jamar & Mejuto (1988) reported that recovery from a small dredge within a harbour was practically complete within 1 year.

In contrast, deep dredge depressions may take longer to physically in-fill, resulting in somewhat prolonged recovery periods compared to shallower trailer furrows and tracks. Dredge depressions in fast tidal streams, for example, exhibited only partial recovery of 57% diversity and 67% biomass after 3 years. On a tidal watershed where water movements were comparatively reduced, a recovery of 85% of species diversity and 40% of the biomass after 4 years was recorded (Van der Veer, 1985).

In general, then, the natural adaptability of the fauna typical of mobile sandy environments means that recolonisation post-dredging is typically rapid and occurs in full. However, this assumes that the nature of the sediment remains similar to that pre-dredging and does not take account of more one-off impacts such as may arise from intensive or static dredging in lower energy environments.

Recovery of Benthos in Shallow Water Mobile Sands

- Recovery of mobile sand fauna commences immediately dredging ceases with some animals settling out from the dredger overflow
- Total benthos recovery is expected to occur within months to 2-4years depending on size and intensity of dredge.
- Recovery of similar species composition will occur if sediment is of a similar composition and structure.
- The rate of recovery is dependant on duration and depth of dredge, hydrodynamic regime, and rate of species recruitment.
- In dynamic environments the rate of recovery is higher than in more reduced dynamic environments.
- The rate of recovery is dependant on magnitude of the impact, with recovery from small operations occurring within 1 year.
- Deeper dredge pits and tracks take longer to in-fill, resulting in prolonged recovery periods.

3.5.2 Recovery in Shallow Water Stable Gravels with Transient Sand.

The initial stages of recovery in gravel habitats affected by transient sand can occur very rapidly, and can even be evident between extractions at a given site (Newell *et al*, 2002). The substantial recolonisation and restoration of community structure within the dredged area within 12 months of the cessation of dredging for such habitats has been recorded. This included the restoration of biomass within the dredged area within the same period. Thereafter, the benthic communities post-dredging are noted as being indistinguishable from those in the surrounding, undredged areas.

This process of recolonisation in these relatively impoverished gravel habitats commences once dredging ceases, although the community that will develop during the initial period after dredging is normally dependent on the nature of the sea bed following dredging. Desprez (2000) concluded that the dredging of gravels in areas of relatively high natural disturbance, due to abrasion, smothering etc from transient sands, may have relatively short-term (circa 3 years) significance because of the rapid recovery of the physical environment and biological communities. In this study, species richness had been fully restored after 16 months with abundance and biomass having recovered to 40% and 25% respectively in the same period.

Similarly, in naturally high disturbance areas, such as those prevalent at the Klaverbank (van Moorsel and Waardenburg, 1991; van Moorsel, 1994), communities are naturally adapted to physical disturbance and this results in a relatively rapid re-colonisation post-dredging, with recovery substantially completed within one year. After such initial recovery has occurred, the attainment of a fully developed more stabilized community may take longer in shallow water areas (5 or more years) as some aspects of the community, particularly sessile and equilibrium species, require several years to reach maturity (Newell *et al* 1998; Rees, 1987).

The naturally occurring transient sand which tends to suppress community diversity in these gravel habitats means that impacts from the deposition of screened sandy sediments are not as significant as in more stable gravel areas (see following section). Nonetheless, these peripheral areas do also exhibit differing recovery rates, from as little as 12 months (Newell *et al*, 2002) to much slower recovery in other studies (Desprez 2000). The variation in these rates would seem to be related to a large degree to the intensity of dredging with more intense, sustained extraction having a longer lasting effect.

In summary, the naturally impoverished nature of these sand scoured or smothered gravel habitats and the adaptation of the fauna to regular disturbance or inundation means that recovery post-dredging to the community that naturally exists can be relatively rapid. This is because the natural fauna is largely composed of opportunistic species with relatively few slower growing, longer lived phyla. What can occur is the inundation of dredge tracks by sandy material which can lead to a small spatial shift in the naturally occurring habitats. Nonetheless, in such naturally disturbed gravel habitats, significant recolonisation and recovery is recorded within 2 to 5 years.

Figure 3.4: *Modiolus modiolus* (Horse Mussel)
Source: Alaska Fisheries Science Center
National Marine Fisheries Service,
National Oceanic and Atmospheric Administration

Recovery in Shallow Water Stable Gravels with Transient Sand

- Initial recovery can be rapid, even between dredging occasions, but generally substantial recovery within 12 months for these habitats has been recorded, including biomass.
- However, the recovery process is dependant on the intensity and frequency of dredging at high intensity extraction site recovery may take longer but is still significantly complete in circa 3 to 5 years, with biomass perhaps taking somewhat longer to fully recover.
- The relatively rapid recovery post-dredging is a reflection of the natural tolerance of the fauna to regular physical disturbance e.g., by waves and tides.
- Secondary impacts from sandy sediment generated by screening or overspill tends to be of low significance in these habitats, primarily due to the natural levels of sand moving over these gravels and the adaptation of the fauna to this instability and regular smothering. However, the significance of these secondary impacts may still vary, relative to dredging intensity and frequency.

3.5.3 Recovery in Stable Gravel

Grab data suggests that rates of recovery in shallow water stable gravel is related to the intensity of the dredging operations in terms of both tonnage and time (Emu, 2002). High intensity, high frequency dredging clearly creates continual disturbance. On cessation of dredging, recovery of the basic habitat and faunal composition in stable gravel areas can occur within 5 to 10 years.

However, studies on these stable gravel areas have indicated the persistence of the physical alteration of the seabed, with the deeply furrowed tracks of the trailer dredgers or the depressions created by anchor dredging still apparent over 10+years post-dredging, although variation in the degree to which they return to a flat and even seabed can be considerable based on the evidence in the available literature.

Figure 3.5: *Ophiothrix fragilis* (Common Brittle Star)
Image: David Connor/ JNCC (Published on MarLIN website)

The majority of the area disturbed, post-dredging, becomes consistent with the previously existing habitat types (as demonstrated by the grabbing studies), including a good mixture of infaunal and epifaunal species. However, the residual areas at the bottoms of the persistent tracks or pits can still comprise a separate habitat type with reduced epifaunal component, due to the deposition of finer generally sandy sediments, which in some cases may become consolidated. A similar conclusion has been reached by Newell *et al.* (1998).

The 'recovery' of a stable gravel community is not, however, as simple as it may appear from these figures as community composition in dredged areas can be altered both within and beyond the dredged areas, reflecting the change in sediment character. Although the initial re-colonising community, generally dominated by polychaetes and juvenile/larval forms, may be regarded as a successional phase in the 'recovery' of an area, the original stable gravel community inhabiting an area cannot re-establish until the physical environment mirrors that which existed pre-dredging.

Screening and overspill operations inevitably lead to the deposition of generally sandy sediment on the seabed. In these stable gravel areas, seabed transport processes may be comparatively slow so that some or all of this sand may persist in and around the dredged area, with slow transport along the tidal axis. This can lead to a significant sandy layer being developed in and around the dredging areas, with bedform features such as sand waves or ripples up and down tide of the dredging. The areas affected in this way will generally be related to the intensity and frequency of the dredging activity, the nature of the resource sediments, the degree of screening employed and the prevailing hydrodynamic regime.

Boyd *et al*, (2003a) found that in an area of historic high dredge intensity, a reduced species assemblage of relatively low abundance was apparent 4 years after the dredging had ceased. The disproportionate occurrence of juveniles within an area subject to high intensity dredging, even after 4 years, indicates that 'recovery' is still at an early successional stage, and that this is likely to be the result of the influence of increased sands in the area. For some areas, then, recovery, or substantial progress towards recovery, may take longer than in others. In the instance of the Boyd *et al* study (2003a), it may be that the relatively slow recovery rate may have been due to the heavy and repeated dredging undertaken within the extraction area studied, coupled with the relatively stable nature of the area as evidenced by the persistence of dredge tracks (7 years).

Although not dealt with specifically in the papers reviewed for dredging impacts, there are instances where more severe impacts and slower recovery will occur. The prime example of this is bio-stabilised sediment, e.g. by the Ross worm *Sabellaria spinulosa*. A loss of sediment cohesion will occur if dredging was to be undertaken in areas where *Sabellaria* have stabilised the underlying gravel sediments and this may lead to a longer-term recovery period in the re-establishment of the 'pre-dredge' community.

In summary, the recovery potential of these deeper water stable gravel communities following the cessation of dredging will depend largely on the nature and stability of the sediment exposed or accumulated at the seabed and the influence of the seabed habitat on the recovery of the pre-dredging fauna. Other factors include the proximity of potential colonizers, both in terms of a population able to migrate and the influx of larval forms; and the degree to which the community is adapted to surviving environmental disturbance.

In general, where the seabed sediments have not been significantly altered, substantial recovery of these stable gravel communities can occur within 10 years, although the slower-growing, longer-lived infauna and epifauna could take even longer in the most stable environments (e.g. the deep water areas of the Eastern English Channel. However, where the seabed has been significantly altered by dredging both within the dredging area and in peripheral zones, for example by the generation of surface sandy sediment features, recovery of the communities that existed pre-dredging will depend on the re-attainment of sediment character and stability and therefore could take substantially longer than 10 years.

Recovery in Stable Gravels

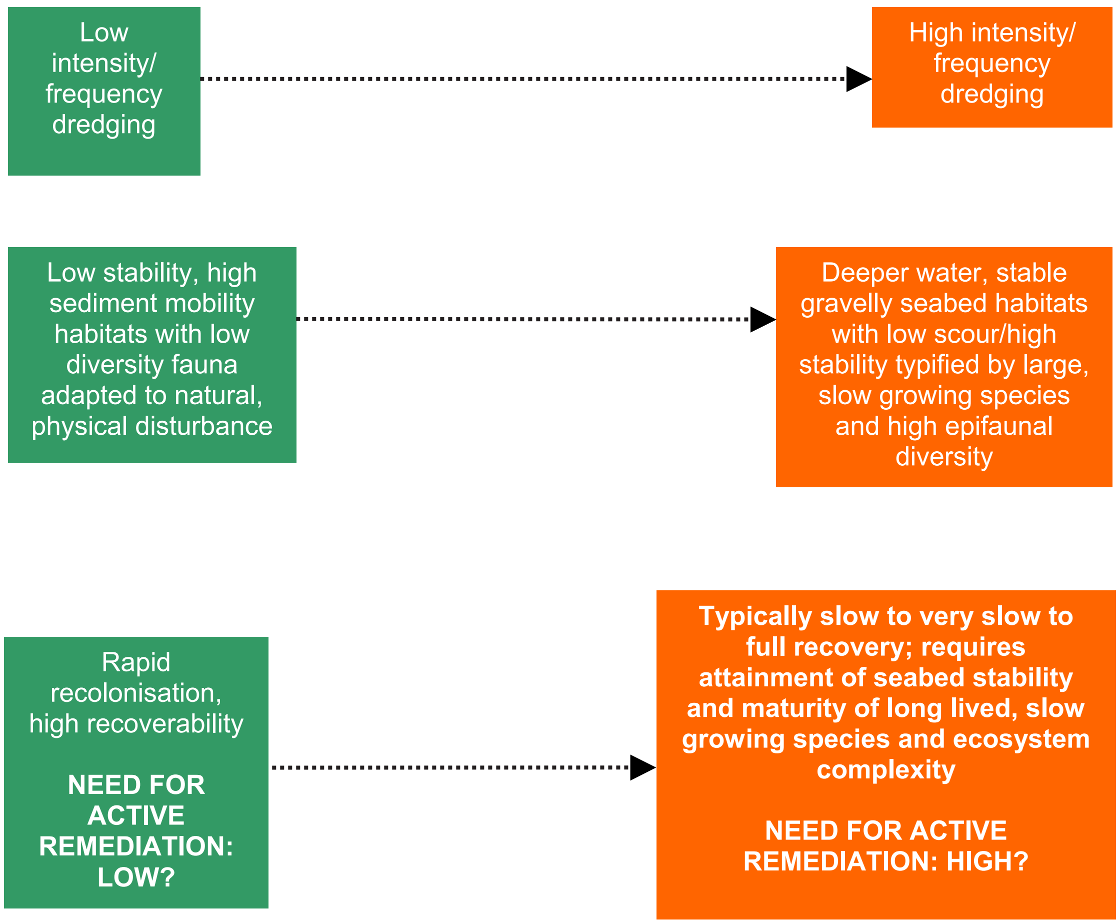
- Data suggests that the recovery rate of these stable gravel areas is related to the intensity of dredging operations in terms of tonnage and time.
- After dredging, recovery of basic habitat and fauna can occur within 5 to 10 years.
- Deep furrow tracks and dredge depressions can persist for over 10 years in some areas. These areas, where filled by finer sediments will be slower to recover.
- Similarly, sandy sediment generated by screening/overspill can change the seabed environment within dredged areas and along the tidal axis, leading to a change in seabed sediments and stability and significantly delaying recovery post-dredging beyond 10 years.
- Deep water gravel recovery will be comparatively longer due to the slower growth rates and less frequent recruitment of some deeper water species and the need to attain suitable, stable sedimentary conditions post-dredging.

3.6 Conclusions

The detailed review of the available scientific literature and industry sponsored research related to the impacts of dredging on the seabed environment has clearly highlighted some key points regarding the sensitivity and recoverability of benthic communities. It is clear that there are a wide number of variables that are linked to the significance of impacts and the ability of the seabed environment and associated faunal communities to subsequently recover. Many of these variables are highly site-specific which means that each site should, appropriately, be considered on an individual case by case basis. This is reflected in the current management practice for many licensed areas whereby the monitoring of seabed habitats and faunal communities is required during and post-dredging.

Nonetheless, in seeking to focus the need for and appropriateness of rehabilitation in relation to marine dredging a number of useful generalisations can be made. These have been summarised in the table at the end of the section.

In order to illustrate the very simplified relationships between the significance and recoverability of the benthos from the impacts of marine aggregate dredging relating to the intensity of dredging and the pre-dredging habitat, the following figure has been developed. As is stated elsewhere in this report, it is considered that assessment of recoverability and the requirement for remediation can only be undertaken on a site-specific basis. There is currently insufficient data in relation to dredging impacts on recoverability on which to make quantified assessments as to where on the continuum remediation becomes appropriate. This dilemma is addressed further in section 6.3 Nonetheless, it is useful in highlighting those marine dredging sites that may take longest to recover and therefore would merit the greatest consideration of post-dredging remediation.



SUMMARY OF NATURAL RECOVERY OF BENTHIC HABITATS

- Impacts on benthic fauna in all habitats are typified by a **decrease in species diversity, abundance and biomass.**
- Recovery of **mobile sand fauna** is expected to occur within a period of **months to circa 2-4 years;**
- Initial recovery of **shallow water stable gravel with transient sand** is expected to occur within 1 year, with recovery of normal community in **circa 5 years**.
- **Stable gravels** are expected to recover within **5 to 10 years**, but with deeper water stable gravels with equilibrium communities taking comparatively longer;
- These rates of recovery are significantly **dependant on the frequency and intensity** of dredging activity.
- Benthic communities typified by **species adapted to periodic natural disturbance** have a **faster recolonisation and recovery rate** e.g. polychaetes. **Sessile epifauna,** especially those associated with stable gravels, are likely to have the **slowest recovery rate**.
- Benthos can only recover to its pre-dredge state once the **sediment composition, structure and stability** replicate that of the pre-dredge site. This is highly significant in stable gravel habitats, but of much less significance in mobile sands and low diversity disturbed gravels where fauna is adapted to low stability and/or high sediment mobility.
- **Recovery rates are slower** where initially stable/consolidated sediments have been destabilised by dredging activities.
- The rate of recovery may also be **slower for persistent, deep dredge tracks and static depressions** in all substrates, due to the time taken for the physical recovery of the seabed. In gravels, dredge tracks and depressions may become in-filled with finer material causing a shift in benthic community.
- **Decreased intensity of dredging** i.e. allowing patches on habitat not directly impacted by dredging, tends to allow for **more rapid recovery** of a site by increasing the availability of existing benthic fauna adjacent to the dredge site.
- **Secondary impacts** from overspill/screened sandy/fine sediments can cause impacts **outside of the dredged area** and tend to be most significant in stable gravel habitats. The extent and impacts arising are **related to dredging intensity, nature of the resource, prevailing hydrodynamic regime** and the resilience of the naturally occurring fauna. Conversely, increased productivity outside of dredged areas has also been recorded, related to the plume of organic material that may be generated by dredging.

4.0 A Review of Existing Methods & Approaches to Environmental Remediation

4.1 Introduction

In this section a definition of remediation is given and the range of approaches which it encompasses described. A review of the literature has highlighted common principles which apply to any form of remediation, and these are discussed in section 4.3. Where appropriate, more detailed discussion has been included in the Annex C and cross-referenced in the text.

4.2 Definition of Remediation

Before going on to consider the existing approaches to remediation, it is useful to establish the accepted principles which underpin the ethos of environmental remediation.

For the purposes of this study remediation is defined simply as '***the action taken at a site following anthropogenic disturbance to restore or enhance its ecological value***'. This is a broad definition which may encompass one or more of the following management options:

- **Non Intervention.** Under this option, the process of natural re-colonisation is allowed to proceed without intervention by man. A distinction needs to be made here from a 'do nothing' approach under which management of the site is either relinquished or restricted to monitoring only. Non-intervention however, might include a regulatory measure to exclude other activities such as trawling to within a certain distance from the remediation site.

- **Restoration** means returning an ecosystem to a condition (or the successional state it would have had) had no disturbance occurred.

- **Rehabilitation** means replacing some of the ecological features of the pre-disturbed system. This is not restoration and is exemplified by replacing mixed forest with strip pine or using landfill / fresh topsoil to allow agricultural use at a former quarry.

- **Habitat Enhancement / Creation** where the original ecosystem or features of it are replaced by a different ecosystem. The scope here is very wide but common examples include the creation of wetlands from former quarries and the creation of artificial reefs.

The above definitions have been drawn in part from those adopted by the National Academy of Sciences (NRC 1992 cited by Pratt 1994). These may be considered primarily as guiding principles and whilst they are certainly helpful in developing strategies for both industry and regulators, experience has shown that complete restoration is in fact rarely achievable and remediation is often a blend of partial restoration and habitat creation/enhancement. In reality the difference between speeding up natural colonisation and habitat creation is blurred, and where the reclamation seeks to build on the inherent characteristics of a site, a decision has to be taken as to the stage at which the natural cycle of colonisation and succession should proceed (Wardell Armstrong 1996).

SUMMARY OF REMEDIATION DEFINITION

- Remediation is the action taken at a site following anthropogenic disturbance, to restore or enhance its ecological value.
- Remediation can encompass a range of approaches from non-intervention through to habitat enhancement or creation.
- Complete restoration of a habitat is rarely achievable.

4.3 Principles of Remediation

For site remediation, as with other forms of environmental management, it is important to establish a set of principles with which to guide the decision-making process on a site specific basis. The review of the existing literature on remediation has identified a number of key guiding principles as follows:

- Remediation is a **complex management issue** (particularly in the marine environment).
- Remediation will always have an **environmental impact**.
- Where possible, remediation should **form part of the planning application**
- The **goals and objectives** of a remediation scheme must be a means to an end rather than an end in themselves.
- The **performance** of remediation schemes needs to be monitored with appropriate corrective measures in place where the goals / objectives are not being met.

These core principles are developed in the following sections.

4.3.1 Remediation Is a Complex Management Issue

In considering any remediation plan, there is a need to define the boundaries of the ecosystem under consideration and this would certainly apply to the marine environment. Ecosystems, by their very nature, are hierarchical open systems which interact ultimately within the global ecosystem. To consider such an open system presents an almost impossible task when seeking to manage particular activities and locations. In furtherance of a practical and manageable approach, an 'artificial' boundary must be drawn around the natural and anthropogenic elements being considered. Such key management decisions must be guided by the existing scientific knowledge (in itself often a considerable constraint).

Where conceptual boundaries are drawn too wide, it can result in a range of scientific, economic and political constraints that become increasingly complex and unmanageable. Defining conceptual boundaries too narrowly fails to address all the significant issues in relation to a remediation scheme, often with disastrous results. There is a catalogue of early remediation schemes for quarries where restoration of the superficial landscape and topsoil was undertaken using landfill without consideration to the problems of groundwater contamination (see paragraph D3 of Annex D). Examples of how the definition of such conceptual boundaries has been attempted are given in Annex C section C.1.

4.3.2 Remediation Will Always Have an Environmental Impact

It is important to recognise that the act of site remediation will, in addition to the positive impacts it may yield, incur a risk of producing significant negative environmental impacts beyond those of the original activity. It is essential that the potential environmental costs of any remediation process are understood, and are assessed against the potential benefits before the decision is taken to proceed.

A responsible approach to remediation seeks to minimise the negative environmental impacts of a given scheme, while maximising the benefits to society within the prevailing technical and cost constraints. Since remediation in the marine environment remains very much a pioneer technology, valuable lessons should be drawn from the terrestrial quarrying industry, whose good practice has evolved over many years, if the remediation of marine sites is to realise its full potential benefits.

It is now widely recognised that the environmental impact of any proposed remediation scheme needs to be assessed. While an EIA is not generally a statutory requirement for remediation, any deposit of material on the seabed will require FEPA consent in advance and probably an EIA. Irrespective of the any statutory requirement all aspects of a normal EIA process should be undertaken, including baseline environmental surveys, consultation with stakeholders and the assessment and mitigation of impacts.

4.3.3 Remediation Should Form an Integral Component of the Planning Application

One important, well established principle developed in the terrestrial mineral extraction industry is that planning for the remediation of quarries should form an integral part of the planning application. In this way, habitat creation can be considered from the outset of a project. This approach is underpinned by planning legislation: schedule 5 of the 1990 Town and Country Planning Act (Schedule 3 of the 1997 Act in Scotland) empowers mineral planning authorities (MPAs) to impose **aftercare conditions** on the grant of planning permission in relation to land which is to be used for agriculture, forestry or amenity following mineral workings.

There would appear to be a clear case for the 'aftercare' of marine aggregate dredging sites being evaluated as part of the initial EIA process. An environmental management program developed through this process could subsequently be designed to monitor the site and address uncertainties, leading to the appropriate and proper application of remediation on a site specific basis.

4.3.4 The Goals and Objectives of Remediation for a Site must be Clearly Understood

Remediation is a means to an end and not an end in itself, and it is important that both the goals and objectives of a project are established and clearly understood. It is incumbent upon industry, regulators and wider stakeholders to ensure that activities in relation to site restoration are ecologically and economically sound.

Several factors should be taken into consideration when setting the goals and objectives for remediation of a site.

Government Policy

Government policy relating to both terrestrial and marine minerals extraction have been reviewed in section 2. Within the key policy guidance several references to site restoration are included which should determine both the need for and approach to remediation schemes (see section C2).

A Thorough Understanding of the Ecosystem

Establishing the goals of remediation may require improved understanding of the original ecosystem and an estimation of the future stability of the site. Baseline environmental monitoring in advance of development increases understanding of ecosystems and their ability to resist, and therefore recover from, environmental disturbance and stress. It also allows the existing value of a site for nature conservation to be assessed.

Performance Assessment Criteria

The objectives should be measurable qualitatively, and if possible quantitatively, and prioritised, since without such criteria the success of a project cannot be assessed. The development of Ecological Quality Objectives (EcoQOs) for marine aggregate dredging sites may prove valuable in this respect, in determining both the need for remediation and the success of the approach (see section 4.3.4 below).

Community Involvement

Community involvement has formed a central guiding principle for many terrestrial schemes. Remediation offers the scope to achieve a whole range of different objectives within the primary goal of restoring/enhancing the ecological value of a site. These can often benefit the wider community as well as the industry itself (e.g. support for education or research into habitat restoration directly benefits the industry, helping it to develop schemes that are acceptable to planning authorities). Community involvement and consensus building can provide further gains beyond the direct socio-economic benefits of the development. The involvement of key marine user groups in the application and design of any management schemes for marine aggregate dredging sites provides transparency and should be given a high priority.

4.3.5 The Performance of Remediation Schemes Needs To Be Monitored

Remediation has been defined as action taken to restore or enhance the ecological value of a site, but implicit in this definition is the existence of criteria to measure ecological value. This presents particular difficulties in the marine environment where there is an incomplete knowledge of ecosystem processes and biodiversity values. Furthermore, only limited information is available on the biological condition of areas originally licensed some decades ago, whilst other marine activities on some parts of the UK continental shelf may have resulted in impacts which predate many marine aggregate workings and continue as an in-combination effect today. This makes the measurement of impacts difficult, and complicates the design and assessment of remediation targets.

Recent emphasis in terrestrial remediation has been on assessing habitat creation schemes in terms of their biodiversity, where species diversity and rarity are key indicators. Plans are frequently linked directly to targets for habitat creation (in terms of spatial extent) or for species populations or biodiversity.

In relation to the marine environment and dredged sites, targets within the UK Marine BAP are currently somewhat sparse, and their direct application to remediation schemes has yet to be determined. There is also a bigger question as to whether biodiversity should be the principal objective of remediation for such marine sites, or whether other criteria such as ecosystem productivity values would be more suitable. A set of EcoQOs for marine aggregate sites may ultimately offer a preferred approach. These could incorporate components of both biodiversity and productivity, and provide an accessible measure of both the need for remediation and a marker against which to assess performance. These objectives could take various forms e.g.

- **Species specific** - restoration of the population of keystone species x within time y;
- **Functional** the sediment is required to support an adequate biomass as fish food (<x% area should have biomass of y gm^{-2} judged against sediment type, salinity and depth) or
- **Based around paired sampling of disturbed and reference sites** e.g. a disturbed site should have recovered to within 80% species diversity and biomass as compared to a reference site within time y.

Extensive treatment of this subject can be found at Cefas (2001).

Failure to meet predetermined targets or objectives in a remediation scheme will signal that a change in approach is required. This subject is discussed in greater depth in section 6.3.2.

SUMMARY OF REMEDIATION PRINCIPLES

Remediation needs to be based upon the following **guiding principles:**

- **Remediation is a complex management issue.** Scientific knowledge can guide the process but managers need to determine the natural and anthropogenic factors to be taken into consideration for a given project.
- **Remediation will always have an environmental impact,** and the potential costs and benefits of a scheme must be understood and evaluated before proceeding.
- **The goals and objectives of remediation must be understood,** taking into consideration:
  - Government policy
  - Understanding of the ecosystem
  - Performance assessment criteria
  - Community involvement
- **Where possible, remediation should form an integral part of the planning Application,** the aftercare of marine aggregates sites being considered as part of the EIA process.
- **The performance of remediation needs to be monitored.** Determining criteria against which to measure performance for marine remediation schemes is problematic, but the development of EcoQOs for marine aggregate sites may offer a potential solution.

5.0 Existing Approaches to Remediation

5.1 Introduction

Little if any remediation appears to have been undertaken at marine dredging sites, so an approach to this subject must draw from existing applications and experience in related areas and industries. The terrestrial quarrying industry has a mature and structured approach to nature conservation which has evolved over the past three decades. This source has proved valuable in drawing out key themes and principles which can shape policy for marine remediation. Coastal remediation, together with applications from other marine industries, offer a fairly extensive inventory of techniques some of which potentially have direct application to marine aggregate site remediation. sections 5.2-5.4 review the existing approaches to remediation drawn from the terrestrial, coastal and marine environments. In undertaking this review, the aim has not been to provide detailed technical descriptions of remediation methods, but to draw out key themes and potential applications for consideration in marine aggregate extraction site remediation.

Where possible, examples have been illustrated with case studies drawn from both the UK and overseas, which aim to show their feasibility and potential impacts if used in aggregate site remediation. If appropriate, more details have also been included in the Annexes C-F, and cross-referenced in the text. The strands of this discussion are brought together in sections 5.5 and 5.6, which, in turn, discuss the applicability of various techniques and offer generic guidance to marine aggregate site remediation.

5.2 The Terrestrial Model of Site Management

Notwithstanding the clear differences in natural environments and scale (see section 2.3) between terrestrial and marine aggregate extraction, there are important parallels which merit consideration in this study. Valuable lessons can be drawn from the remediation of land-based mineral extraction sites which can contribute to the development of remediation policies for marine aggregate dredging sites.

The aim of this section is not to provide a detailed review of remediation approaches and techniques in the land-based quarrying industry, but rather to focus on the common themes to inform an approach to remediation in the marine environment. This review has highlighted three themes which are considered in some detail in Annex C section C3 and Annex D, but may be summarised under the following:

- Physical similarities between the terrestrial / marine aggregate sites;
- Similarities in the principles of ecological succession;
- Environmental impact.

5.2.1 Physical Similarities between Terrestrial / Marine Aggregate Sites

The removal of overburden to access underlying aggregates is a feature of quarrying which can also occasionally apply to marine aggregate sites. Where the objective of remediation is to restore to the pre-quarried state, loss of overburden can extend, if not completely impede, the recolonisation process.

Quarrying and dredging can result in the creation of atypical landforms. Former quarries can comprise a whole range of features including mounds, hollows, pools, ledges and ridges which can provide important habitat niches (Wardell Armstrong 1996). There is some evidence to suggest that dredging can occasionally result in the creation of habitats which are of ecological interest and value (see Annex F section F2).

The features created by both quarrying and dredging can be inherently unstable. For the marine environment safety is not an issue as it is on land – although dredging can expose fasteners (snagging hazards) which can present a safety hazard for bottom trawlers. The instability of both dredging depressions and natural sinks for screening material can have an impact on the surrounding habitats.

5.2.2 Similarities in the Principles of Ecological Succession

The general principles governing community structure following environmental disturbance appear to be generally applicable to a wide variety of communities, both on land and on the sea bed (Newell *et al.* 1998). There is value to be gained from reviewing how the principles of ecological succession have been harnessed in remediation of terrestrial sites, and applying these concepts to marine aggregate sites.

5.2.3 Environmental Impact

Remediation, whether for quarries or marine aggregate sites, will always have an environmental impact. While not a statutory requirement, the EIA process forms an integral part of many remediation schemes for quarries. Best practice in this area, cultivated over several decades in the quarrying industry can help to shape remediation approaches for the marine environment.

5.2.4 Approaches to Remediation in the Terrestrial Environment

The review of the terrestrial experience has highlighted four main approaches to site management following quarrying activity, which broadly follow the core themes for remediation and management presented in section 5.2. A detailed review of examples of these various approaches is presented in Annex D.

Non Intervention / Natural recolonisation

There exist a large number of historic pits and quarries where excavation ceased many years ago, in which the natural processes of colonisation were allowed to proceed without interference, and many of these now support habitats analogous to those on virgin land. Initial colonisation would probably have occurred from adjacent undeveloped land on the same substrate. Whilst this seems to be an attractive proposition at first sight, non-intervention is rarely an option if nature conservation objectives are to be met at terrestrial sites. Habitats are often likely to be colonised by a different range of species than those once typical of the substrate.

Intense agricultural and urban development has transformed the terrestrial landscape and removed many areas of natural habitat. Pressures on the seabed tend to be of a different nature. Colonisation by alien or undesirable species and degradation of the gene stock is a recognised concern in areas of intensive mariculture but a more significant impediment to natural recolonisation at marine aggregate sites may be the lack of measures to restrict cumulative disturbance from trawling / scallop dredging. Marine Protected Areas, discussed in section 5.4 below may offer one potential solution.

Figure 5.1: Naturally Recolonised Limestone Quarry, Stroud, Gloucestershire. Source: Kids Ark

Restoration of Ecosystems

Complete restoration of an ecosystem to a pre-excavation state in the terrestrial environment is not a feasible option. Since excavation removes huge quantities of substrate, which cannot be put back, inevitably this affects the fundamental characteristics of the ecosystem, the nature of the underlying soil and rock, drainage, nutrient supply, sunlight etc. and the resulting flora and fauna. Conversely, it is the very rarity of some of the naturally recolonised quarries, which has made them targets for nature conservation. It is often the early colonisation species which are particularly significant. Left unmanaged, invasive species will begin to threaten and dominate.

In the marine environment, restoration to a condition approaching the pre-excavated state is likely to be more attainable, since the habitat adjacent to excavation sites is often not dissimilar in character[1], and the key processes of weathering, sunlight exposure and groundwater flow which form habitats in the terrestrial environment do not apply to the same degree. Recognising that the species composition of communities is dynamic, Newell *et al* (1998) suggests that a practical approach to "recovery" will be the recognition of the establishment of a community that is capable of maintaining itself, and in which at least 80% of the species diversity and biomass has been restored.

Topographical Rehabilitation

Topographical rehabilitation in the context of this study is using material to fill an excavation once it has been worked. There are important lessons that have been learnt from early experience with this form of remediation, which are of direct relevance to recent policies to use former static dredging or borrow pits for dumping waste (See sections 5.4 and D.3). Historically, the use of land-based quarry sites from land fill has led to extensive environmental problems unforeseen at the outset, and which have resulted in subsequent expensive remedial action. Such cases serve to illustrate the problems attendant with remediation schemes which fail to adequately address the environmental impacts of a proposed scheme on the wider ecosystem.

Habitat Creation

It is evident that nearly all forms of remediation involve some form of habitat creation or enhancement, albeit that the objective may be to recreate former habitats. There is a distinction between the approaches outlined above in which the focus is habitat management of the baseline ecological conditions, and habitat creation schemes which specifically set out to develop a habitat for a species or community of species, which did not previously exist at the site. This approach forms a specific aim of government policy for land based sites:

'*Extraction of minerals can create new types of habitat in areas where they were formerly rare or absent, while quarry faces may provide a valuable supplement to natural rock outcrops since geological features may be revealed during quarrying operations.'*

In the terrestrial environment, habitat creation has several important roles:

- contributing to increasing biodiversity and the achievement of BAP targets;
- linking fragmented areas of semi-natural vegetation in an otherwise intensively used landscape;
- buffering existing semi-natural areas;
- expanding existing sites, so making their management more practicable or economically viable;
- providing new areas which can serve as a local amenity and educational resource, and which can draw people away from fragile habitats.

It may be that the selective creation of habitats at marine dredged sites could serve to address some of the same ecological and biodiversity objectives, which form the basis of many terrestrial schemes.

[1] The similarity of adjacent habitats can, in part, be attributable to licensing policy, for example zoning within licence areas which seeks to minimise disturbance to the seabed at any one time.

SUMMARY OF TERRESTRIAL REMEDIATION

- A **non-intervention** approach is rarely an option for terrestrial sites, due to the problem of colonisation by invasive species from adjacent areas. Conversely, the more homogenous nature of the seabed allows natural recolonisation by desirable species from adjacent habitats.

- **Restoration** in former quarries is frequently targeted on early colonising species in ecosystems which, have to be actively managed to prevent domination by invasive species. In the marine environment, it has been suggested that a practical approach to recovery, would be to achieve a viable community in which at least 80% of the species diversity and biomass has been restored.

- **Topographical restoration** using land fill to landscape former quarries has caused environmental problems, due to a failure to adopt an ecosystem approach. This experience has relevance to filling of dredging depressions.

- Exploiting the opportunity for **habitat creation** at former quarries is a stated government aim to promote biodiversity. Selective habitat creation at marine dredged sites could address similar biodiversity objectives.

5.3 The Coastal Model of Site Management

The management of coastal habitats is an increasingly prominent consideration in coastal zone management. Habitat creation or restoration at coastal sites is undertaken using a variety of techniques to achieve a range of objectives including the following:

- Coastal defence and flood alleviation;
- Groundwater recharge;
- Beneficial use of dredged materials;
- Archaeological conservation;
- Shore stabilization;
- Fishery and shell fish production.

However, currently the majority of coastal habitat creation and restoration schemes have the primary objective of providing habitat for nature conservation, either in terms of mitigation as the result of habitat loss through development, or as a result of natural habitat loss or degradation through erosion.

A number of examples of coastal sites around the UK, historically used for aggregate extraction where management has been attempted are reviewed in Annex E.

5.3.1 Coastal Aggregate Extraction

Extraction of sand and gravel from coastal structures, typically shingle beaches, has caused structural deformation and loss of vegetated shingle habitat. In some cases, restoration has been attempted to restore the original shingle habitat in terms of its particle characteristics and topography in order to encourage the recovery of the associated flora. However, although recreation of habitats has been partially successful, restoration of the associated vegetated shingle was not achieved, at least in the short-term.

More successful attempts to restore sites affected by aggregate extraction have been those parallel to the terrestrial models reviewed above. For example coastal shingle extraction sites have been turned into lagoon and wetland habitat with considerable success. This has helped to contribute to regional BAP targets and provided habitats for rare plant and animal communities.

Figure 5.2: Restored Coastal Aggregate Site, RSPB Reserve, Dungeness, Kent.
Source: Hanson

5.3.2 Management of Intertidal Habitat

The management of coastal intertidal habitats, for reasons not directly associated with the aggregates extraction industry, is an increasingly important management priority in many coastal areas. Such management can be achieved through several processes including:

- Coastal realignment;
- Enhancement sedimentation;
- Foreshore recharge.

Of these, coastal realignment is not applicable to the management of marine dredged sites and will not be considered further, although much literature is available (e.g. ABP, 1998). However, the remaining two techniques may provide some useful guiding principles and examples in considering the management of subtidal areas.

In the case of enhanced sedimentation the primary aim is to manage the natural processes acting on intertidal habitats, in order to encourage active sediment accumulation, and slow or reverse the processes of erosion. A number of such projects have been attempted in the UK using a variety of techniques (see Annex E). Some of these projects have proved successful in increasing the rate of sedimentation, but generally have been found to work over long timescales (which may be appropriate in such circumstances) and generally only where there is a tendency for natural accretion at the coast. The successful restoration or creation of associated sensitive habitats such as saltmarsh, even where the level of the mudflats has been raised, is not always recorded. This tends to reinforce the difficulty of large-scale habitat management in such complex systems, and in the face of the natural processes which act on them.

Foreshore recharge aims to utilise sediment, often from capital or maintenance dredging projects, to manage intertidal and saltmarsh habitats. The main aim is to maintain the surface elevation of these habitats against natural processes, and combat longer-term erosion. Several methods can be used to place the material on the intertidal area, and a number of such projects are reviewed in Annex E. Many of these projects have been successful in their long-term aims of maintaining or raising mudflat heights, and even in encouraging the growth of saltmarsh vegetation. Generally, these projects resulted in an initial impact on the intertidal fauna as a result of smothering. This was followed by subsequent re-colonisation of the fauna, typically of the fast growing opportunistic species. However, it is noted that many of these projects were relatively small-scale and served quite specific habitat management aims.

Figure 5.3: Foreshore Sediment Recharge.
Source: BMAPA

Work undertaken in coastal habitat remediation has direct relevance to marine remediation. Physical processes which govern enhancement sedimentation apply also to sedimentation in relation to dredging. In particular, one of the potential benefits of the proposed scheme to fill ‘borrow pits’ in New York / New Jersey harbours, is an improvement to the local sediment regime (see paragraph F.3). Similarly, pioneering work in topographical restoration of former dredging sites, can draw from experience gained in foreshore recharge and its effect on the benthos to some extent.

SUMMARY OF COASTAL REMEDIATION

- Coastal habitat restoration / creation is undertaken to achieve a variety of objectives, with nature conservation being a primary goal.
- Remediation of shingle beaches depleted through gravel extraction has been attempted, but restoration of vegetated shingle has not always been achieved.
- Former coastal shingle extraction sites have been successfully used to create lagoons and wetlands, and to help meet BAP targets.
- Intertidal restoration schemes including sediment enhancement and foreshore recharge provide an important knowledge base upon which to develop pioneer work in marine remediation.

5.4 The Marine Model of Site Management

Unlike the terrestrial minerals extraction industry, the literature review has indicated that there are few, if any, direct examples of remediation of marine aggregate dredging sites, either in the UK or elsewhere, which could form the basis of an approach to remediation policy for marine sites. Rather, existing measures are in place to minimise the environmental impacts of aggregate dredging which will ensure the recovery process is less likely to be impeded (see section 2.5.2).

There are a range of remediation methods that have been applied outside the aggregates industry under a variety of scenarios, predominantly outside of the UK, which may provide a suitable starting point for policy development. Many of these methods and approaches for marine site management are similar to those used in the terrestrial and to some extent coastal areas, and therefore continue some of the core themes established in section 4 and developed through sections 5.2 and 5.3.

5.4.1 Non Intervention / Natural Recolonisation

The seabed, unlike the terrestrial environment, has not been subject to the same degree of extensive development or interference. In comparison with terrestrial habitats, marine ecological communities tend to be more resilient to stress due to the natural dynamics of the marine environment (see also section 2.4). Furthermore, broadly similar seabed habitats supporting similar faunal communities often exist over large areas and larval recruits may travel very large distances, so that post-dredging re-colonisation will often occur in the short to long term. The rate and extent of recolonisation is instead more directly related to the sediment composition and stability at the dredged site, and the nature of the prevailing hydrodynamic regime.

A non-intervention approach relying on the natural recovery process (but often with a monitoring program in place) is essentially the management option currently employed at all existing and historic marine aggregate dredging sites.

5.4.2 Active-Passive Recovery

A further option beyond the simple non-intervention approach is the designation of non-disturbance zones, or marine protection areas (MPA) at relinquished dredging sites. This would exclude all other marine development and fisheries activity, removing any residual impact on the site. Such MPAs have been explored in other countries and the benefits to fish stocks and development of eco-tourism have been noted, particularly as a result of excluding fishing activity. These projects have proved to be most useful on naturally stable areas not normally subject to disturbance; at naturally disturbed sites, they are likely to be of less benefit.

Another alternative may be the incorporation of an exclusion zone more akin to those utilised by EU, such as Fisheries Exclusion Zones (FEZs) or Closed Areas. These FEZs have been used on a huge scale in the North Sea and around the UK coast with the Plaice Box in the Eastern North Sea, where towed fishing gear is prohibited, covering an area of approximately 38,000km^2. A striking example of the success of a closed area is the Georges Bank, in North Eastern USA, where the number of scallops increased 15-fold inside an area closed for 4 years.

Notwithstanding their potential benefits, FEZs and MPAs would clearly have negative impacts, particularly on trawling vessels and other marine development activities. Their introduction is likely to be contentious and there would need to be very clear quantifiable benefits to justify their use. Field trials would be necessary to compare recovery rates of sites both with and without the designation of non-disturbance zones. If such trials were to demonstrate a significant reduction in the period of recovery post dredging, then they may offer a viable remediation option. With any regulatory measure of this nature, there is an additional issue of enforcement. If such measures are not self-policed, they can prove extremely difficult and prohibitively costly to enforce. Clear demonstration of their value from field trials together with appropriate consultation would probably increase the viability of a self-policing option.

5.4.3 Preferential Use of Marine Aggregate Sites

There may exist a case for designating former aggregate extraction sites as preferential sites for other marine activities, for example windfarm development. Clearly all other criteria for windfarm site selection would need to be satisfied but use of former aggregate extraction sites has potential dual benefits. It would ensure that construction disturbance associated with windfarm development would be concentrated in an area which is already likely to be faunally impoverished; furthermore, long-term recovery of the benthos and pelagic marine life may conceivably benefit from the absence of the exclusion of bottom dredging / trawling activity by virtue of the wind turbine structures.

5.4.4 Conserving the Modified Seabed

There is some evidence to suggest that dredging activity, particularly that associated with static dredging, can sometimes create a distinctly different habitat from the surrounding area, in which the changes in the topography of the seabed can provide micro niches for certain species which are of enhanced biodiversity value. For example, video and still camera surveys of the former static dredging depressions on the Inner Owers Bank have found the base of the depressions to be colonized by dense communities of *Mytilus edulis* (common mussel), together with the branching bryozoan *Pentapora foliacea*, constituting habitats which are very distinct from surrounding sand and gravel habitats (see section F2).

As policy is developed to consider what remediation, if any, should be undertaken at marine aggregate sites, further investigation should be made into the potential intrinsic value of some former dredging sites, particularly if increasing biodiversity in the marine environment emerges as a core theme as it has for the management of terrestrial sites.

5.4.5 Marine Site Restoration

Restoration is probably the approach most commonly associated with remediation. However, subtidal restoration in the UK appears to be very much in an early experimental stage. One site at which restoration has been planned and undertaken is the Roughs Tower dredging disposal site offshore of Harwich. The site has been used for disposal of capital dredgings associated with the expansion of the port of Felixtowe. The placing of a clay bund around this disposal site, together with gravel seeding formed a mitigation package for the 1998 capital dredge (see section F3). Marine site restoration is under extensive consideration by the offshore oil industry in relation to remediation of drill cutting piles. The consultation BP have currently entered into in relation to decommissioning of the West Hutton oil rig provides some transferability in approach which may inform the debate on restoration of dredging sites (see section F3.2). Gravel dumping is used routinely for armouring pipelines, but could also be used to create favourable habitats. A major project in New York Harbour involving **topographical restoration** of former borrow pits (see section F3.3) has particular relevance to dredging sites. Potential benefits include habitat restoration, an economic site for dredging disposal, and possible improvements in water quality. However, potential environmental risks have also been identified including the loss of the unique habitats in the pits, possible water quality impacts through the disturbance of contaminated sediments, and possible unforeseen hydrodynamic impacts.

A common theme for all of the existing examples is the process of **capping**, covering up an impacted area of seabed with a preferred sediment layer, to entrap contaminated material and/or to encourage the recovery of a more natural biotope. This process has some potential for deployment at marine dredged sites, where sediment deposition could be used to restore the physical condition of the seabed, thereby speeding up the natural recolonisation process. Based upon the experience with capping of a historic contaminated disposal site in New York Bight, the quantities of material required to cap a relatively small area are immense (an estimated 40 million m^3 of dredged material will be required to complete a capping scheme over a site covering 2.2nm^2)(see section F3.4). To set this in context, this broadly equates to 60million tonnes of aggregate which represents 3 - times the current annual marine aggregate production. On this basis, the volumes of material required for capping rule out its feasibility except for very localised sites.

Gravel dumping is an established capping technique used in the offshore industry for applications such as adding a protective cover to exposed or free-spanning pipelines. The method involves dumping material ranging in size from gravel to small boulders from surface vessels; gravel is thought to be preferable due to the reduced hindrance to commercial trawling. The method is known to smother benthos living in or on the sediments. It has also been considered as part of a package for remediation of drill cutting piles.

Furthermore, a review of the existing studies on marine restoration also provides some cautionary evidence. For example, at the Harwich site mentioned above, attempts to manage lobster colonisation through hatchery ranching have not proved highly successful, although natural recolonisation has occurred in the longer term. This example perhaps provides an indication of the risks of 'ambitious' interventionist policies, without a highly robust knowledge of the ecosystems involved.

5.4.6 Marine Habitat Creation / Enhancement

The distinction between habitat restoration and habitat creation and enhancement for the marine environment is not clear-cut, and differentiation is more a factor of management objectives than methods. A range of existing examples have been identified which fall into this broad category of marine environmental management, including the seeding of gravel or shells in order to improve the seabed substrate, notably for shellfish harvesting areas, and the use of artificial reefs and fish aggregating devices. A number of case studies of these various techniques are included in Section F.4. Artificial reefs are considered separately below.

The various habitat creation or enhancement initiatives tend to focus on either;

- Promoting particular habitat features which are broadly consistent with the existing seabed environment around the development sites in question, often with a particular aim such as promoting the development of sustainable fisheries.

- Or enhancing the habitat diversity of an area in order to increase productivity and/or biodiversity, principally by the use of artificial structures.

Both of these aims could be applicable to the management of aggregate dredging areas but both also have potential practical, political, and environmental issues associated with them.

The placement of shell-gravel to encourage particular biotopes or species has been attempted in several areas in the USA, specifically for oysters and clams. Typically shells of the target species were spread onto the target areas of seabed (or in some cases reefs to encourage oyster settlement), with reported notable increases in the productivity of the fishery targeting the particular species involved. In New Zealand the spreading of scallop shell onto the seabed to encourage scallop settlement is under discussion, and could represent a possible remediation measure at some UK sites. The results of these various studies tend to suggest an increase in both the target commercial species and the diversity of the associated epifauna, as well as helping to armour and stabilise the seabed sediments.

Figure 5.4: Oyster Shell Dumping, Raritan Bay, New Jersey, USA.
Source: Baykeeper webpage, New Jersey

However, in applying such shell-gravel seeding as a management technique for aggregate dredging areas, a number of key guiding principles should be considered:

- Its use should be confined to aiding the recovery, or promoting the productivity of naturally occurring shellfish populations. Introduction of such species where they do not currently exist would risk possible impacts on the biodiversity of dredged areas and disruption of existing ecosystem processes.

- Stakeholder consultation (particularly with the fishing industry) should be a prerequisite.

- Schemes (as for all remediation projects) must be subject to detailed cost benefit analysis.

- Such schemes may be best applied regionally rather than at every site, one or two such schemes in a region affected by dredging may be more manageable and successful than when applied over large areas.

Nonetheless, it is considered that where environmentally applicable, the process of shell-gravel seeding should be considered, given the apparent importance of such material in the benthic ecosystem processes of natural shellfish habitats. This could also provide an environmentally beneficial way of using some of the waste produced by the shellfish processing industry.

5.4.7 Artificial Reefs

An artificial reef is one or more objects of natural or human origin deployed purposefully on the seafloor to influence physical, biological or socioeconomic processes related to living marine resources. Items used in reef construction provide a vertical profile to the benthic environment, and may be either assembled expressly as a reef or acquired after being used for another purpose e.g. scrap ships.

A key objective for an artificial reef is to provide a habitat different to that of its surroundings, and in that sense to enhance the biodiversity of an area as well as increasing productivity. Against this, the potential impacts upon the intrinsic ecology and biodiversity at the site need to be carefully considered. There exists a danger that increased biodiversity *per se* can be used to justify artificial reef deployment to the exclusion of considering the intrinsic native ecology. Any new structure placed on the seabed is likely to have an impact on the existing ecology, most noticeably by the smothering of the benthic habitat beneath its foot-print. Additionally, changes to the physical processes of the area and introduction of predator species can be to the detriment of an existing community.

In relation to the management and enhancement of aggregate dredged areas, the use of artificial reefs would not always be a practical solution and would be highly dependant on the seabed characteristics, possible impacts to the benthic environment, species composition, and other seabed users. However, this type of habitat creation may be applicable in certain areas, for example where particular conservation aims or fisheries management schemes are applicable, and agreement is reached with relevant stakeholders.

Figure 5.5: Artificial Tyre Reef, Poole Bay, UK.
Source: Dr. Ken Collins, Southampton Oceanogaphy Centre

As for the other types of habitat enhancement reviewed above, it is possible that small artificial reef schemes used on a regional scale could form the basis of a wider management approach, with the aim of enhancing biodiversity or encouraging valuable fisheries. Such reefs, properly managed could provide a useful addition to the protection or enhancement of certain fish and shellfish stocks, with the support of relevant authorities and stakeholders, although the practical issues of design and licensing of such artificial structures are recognised.

However, as with any remediation scheme a cost-benefit analysis and a detailed consideration of the potential environmental impacts would be essential in considering the use of artificial reefs in previously dredged areas

Three further broad categories of habitat management were recorded from the literature review, but were not felt to be applicable to the management of aggregate dredging areas. However, brief descriptions of these are presented in sections F4 (3) to F4 (5), they are:

- Fish aggregating devices (FADs)
- Artificial fertilization of the sea
- Ecosystem modification

SUMMARY OF MARINE REMEDIATION

The following suggested approaches to remediation of marine aggregates sites are based on techniques used in a variety of marine and coastal applications, but also draw key principles from remediation within the quarrying industry.

- **Non Intervention / Natural Recolonisation** is directly related to the sediment composition and stability at a site, and the prevailing hydrodynamic regime. In conjunction with monitoring, it is the current management option for most UK dredging sites.

- **Active-Passive Recovery** relies upon natural recolonisation, with the additional measure of designating a non-disturbance zone / marine protected area (MPA) around a site. This is more suited to naturally stable areas not subject to disturbance. MPAs are invariably contentious.

- **Conserving the Modified Seabed.** Evidence suggests that dredging in some areas has created habitats of some intrinsic value which may be of conservation interest. A review of former dredging sites in this context may prove valuable.

- **Marine Site Restoration** Techniques used in marine industries reviewed including capping and gravel dumping have potential application to aggregate site remediation. Examples of schemes reviewed include:

  Roughs Tower dredging disposal site Gravel seeding to provide a favourable habitat for lobsters and a juvenile lobster introduction programme, the latter having proved problematic;

  Topographic restoration of former dredging pits A proposal to fill pits in New York Bight with maintenance dredging spoils and cap them to allow recolonisation;

- **Marine Habitat Creation / Enhancement.** Differentiation between habitat restoration and habitat creation / enhancement is not clear cut and more a factor of management objective than methods.

  Placement of shell or gravel to encourage particular biotopes or species, merits consideration given the apparent importance of such material in the benthic ecosystem processes of natural shellfish habitats at some UK sites.

  Artificial reefs have proved successful in boosting benthic productivity in degraded habitats. These may be applicable in certain areas within wider management schemes aimed at conservation or enhancing fisheries.

5.5 Applicability of Existing Remediation Methods to Marine Dredging Areas

Remediation method 1. Administrative	Existing	Marine	Physical Scale / Issues
Nature Reserves	• Extensively used to preserve former quarries for conservation / recreational use in increasingly developed landscape. • Can benefit wide cross-section of society • Reserves are easily enforceable. • Management responsibility can be shared by range of stakeholders.	• No record of use for former dredging sites which generally constitute denuded habitats with a reduced intrinsic biodiversity value. Marine reserves are principally used in inshore areas as a component of ecotourism initiatives to preserve existing areas of high biodiversity or for specific conservation aims. • Access problems would restrict the direct enjoyment / benefit to a narrow sector of society principally divers. • The difficulty and cost of policing marine reserves increases with the size of area and distance offshore. • Narrowness of interest groups reduces the scope for shared management. • Could potentially be used in a compensatory context as part of a regional biodiversity strategy.	• Difficult to enforce and manage reserves over a large scale. • Practical difficulties in designating entire licence areas as a nature reserve, due to their size, however potential offshore cSAC may cover similar or large areas of seabed.
Temporary Exclusion Zones / Closed Seasons	• All conservation areas represent exclusion zones for a range of activities and management determines when species can be harvested for recreational (e.g. game fishing) or commercial (e.g. tree felling) reasons. These measures work in conjunction with closed seasons which tend to be universally applicable to sites nationally. • They require active enforcement by rangers. • Their use is generally acceptable to society as a whole.	• Exclusion zones have been used extensively for conservation and protection of fish stocks. E.g. commercial fishing is excluded from a number of estuaries and bays in the UK which are important nursery areas for juvenile commercial species, including River Exe, River Conwy and Filey. Use specifically as an instrument for benthic recovery is not known. • Large scale fisheries exclusion zones (FEZs) have been implemented by the EU, where large areas are closed to fishing of certain key species and/ or certain types of fishing gear are prohibited • Enforcement as with Marine Nature Reserves is a problem and costly. • Their use is frequently controversial because of the vested interests of the fishing industry and transboundary issues.	• Exclusion zones have been used on a large scale by the EU, with the *Plaice Box* (where towed gear is prohibited) covering an area of 38,000km^2. • There is potential for exclusion zones to cover entire license areas, however the enforcement and stakeholder involvement of such zones would need to be considered.

Remediation method 2. Physical Restoration/ Enhancements	Existing	Application to Marine Aggregate Dredge Site Remediation	Physical Scale / Issues
Gravel / shell capping	• Use in remediation has generally been confined to small areas. In Harwich Haven a sprinkling of gravel was applied to an area less than $4km^2$. US army corps of engineers habitat creation projects involving gravel have ranged from 1196 m^2 to $14000m^2$ □requiring deposit of approx. $22000m^3$ of material (USACE 1989).	• Active dredging sites average $12km^2$ suggesting the application of gravel / shell capping on a wholesale basis is infeasible (e.g. 0.5 metres of gravel over $10km^2 = 5\ Mm^3$) • May have application for particular features such as static depressions or for sediment sinks down-tide of former dredged areas. • The source of capping material and the potential impacts in acquiring the resource could be significant and needs to be considered fully.	• It may be infeasible to suggest capping an entire licence area. • Highly targeted capping of specific areas may be a feasible at certain sites.
Physical restoration / profiling	• Use of former quarries as waste landfill sites has proved both convenient and economic. If properly engineered, waste fill can remain isolated from groundwater flow ensuring minimal environmental impact.	• More complex due to the dynamics of the marine environment. Has been applied to borrow sites in Boston harbour and is under consideration for New York former static dredging sites as a convenient and economic means of disposing of harbour maintenance dredgings • Conceivably, could be applied to inshore former static dredging sites in the UK. Economic attractiveness would need to be balanced against the potential environmental impacts.	• Huge volumes of material would be required to topographically restore marine dredge sites. Sourcing these volumes is likely to be impractical, • Targeted restoration of specific small areas may have potential.
Habitat Enhancement	• Artificial reefs have been used extensively for over 3 decades and evidence suggests they can enhance benthic habitat, particularly in areas of low productivity / degraded habitat.	• Offers potential benefit to habitats degraded through dredging but the benefits are very localised and their placement would need to be highly selective, within a wider context □ possibly a marine BAP. • Possibility exists to increase productivity in a localised area, potentially providing new fisheries opportunities.	• Habitat enhancement over entire licence areas may be infeasible. • Targeted use of habitat enhancement measures at suitable sites may be viable to increase productivity and/or biodiversity of an area. • Habitat enhancement may have potential as a component of a regional management plan.

5.6 A Generic Approach to Remediation of Marine Aggregate Sites

Guiding principles for remediation schemes in general have already been identified in section 4. Based upon these, the guidelines set out below are suggested as a generic approach to remediation of marine aggregate sites.

1. Establish the need for remediation;

2. Establish the goals and objectives for remediation of a site;

3. Determine the criteria against which the performance of a scheme will be measured;

4. Adopt an Ecosystem approach;

5. Follow an EIA process:

   a) Scoping;
   b) Consultation;
   c) Collate baseline environmental data;
   d) Identify remediation options and their associated impacts;
   e) Select preferred remediation option;
   f) Write a detailed remediation plan including pre, in-progress and post monitoring programme;
   g) Undertake work;
   h) Evaluation against pre-determined criteria.

Step 1 - Establish the Need for Remediation

Policy guidelines for remediation of marine aggregate sites (reviewed in section C.2) are loosely defined. In the absence of clear regulatory guidance, the need for remediation at existing sites needs to be determined on a case by case basis, drawing upon the best scientific knowledge available.

It has been shown in section 3, that the impacts of dredging vary extensively according to seabed conditions and the intensity of dredging. The development of industry guidelines to determine the criteria for initiating remediation would be beneficial in establishing the 'need' at a given site.

Step 2 - Establish the Goals and Objectives for Site Remediation

Current policy requires the seabed to be left in a similar condition to the pre-dredged state. This would suggest the goal of remediation should be restoration, as opposed to habitat creation or enhancement. Given the dynamic nature of ecosystems, 100% restoration of species composition and/or population may not be realistic, and there exists a need to define 'restoration' in a manner which can be generically applied to marine remediation. For example in relation to static dredging depressions, it is open to interpretation whether topographical restoration is appropriate or not. Secondary objectives which might benefit other stakeholders should also be identified through consultation.

Step 3 - Determination of Performance Criteria

This is an area in which further work is necessary to help develop industry guidelines, although current work on the development of EcoQOs for the industry could contribute directly to this requirement.

Step 4 - Adopt an Ecosystem Approach

There is a need to draw a conceptual boundary around the ecosystem under consideration within any remediation scheme. Scientific knowledge can guide the process, but decision makers need to limit the natural and anthropogenic factors to be taken into consideration. Some are more clear-cut than others. There will always be a need to consider and consult with the local fishing community and to evaluate the potential impact on benthic an pelagic ecology directly impacted. The relevance of interaction with the coastal sediment system for an offshore deepwater site may be less clear.

Step 5 - Follow an EIA Process

EIAs, particularly for smaller schemes such as small scale gravel seeding, may not be a statutory requirement, but experience has shown the benefits of following an EIA process which will ensure that all environmental factors are taken into consideration, and decisions are made based upon the significance of both the positive and negative impacts. Collation of baseline environmental data is essential to determine both the need for remediation, and the datum against which the objectives of a scheme can be measured. Pre, in-progress and post remediation monitoring regimes will allow progress of the scheme to be evaluated.

SUMMARY OF SECTION 5

Remediation is interpreted simply as the action taken at a site following anthropogenic disturbance, to restore or enhance its ecological value. It can encompass a range of approaches from non-intervention through to habitat enhancement or creation.

Complete restoration of a habitat is rarely achievable.

Remediation needs to be based upon the following **guiding principles**:

- **Remediation is a complex management issue**. Scientific knowledge can guide the process but managers need to determine the natural and anthropogenic factors to be taken into consideration for a given project.
- **Remediation will always have an environmental impact** and the potential costs and benefits of a scheme must be fully understood and evaluated before proceeding.
- **The goals and objectives of remediation must be understood**, taking into consideration
  - Government policy;
  - Understanding of the ecosystem;
  - Performance assessment criteria;
  - Community involvement;
- **The performance of remediation needs to be monitored**. Determining criteria against which to measure performance for marine remediation schemes is problematic, but the development of ecological quality objectives (EcoQOs) for marine aggregate sites may offer a potential solution.

The following are suggested as potential **options for remediation** at marine aggregate sites.

- **Non Intervention / Natural Recolonisation** - The current management option for most UK dredging sites.
- **Active-Passive Recovery** Use of potentially contentious marine protected areas to limit disturbance by other sea users and aid natural recolonisation.
- **Conserving the Modified Seabed** May apply at certain former dredging sites where anthropogenically created habitats are deemed of intrinsic biodiversity value.
- **Marine Site Restoration** - Applying a range of techniques including capping, gravel dumping or species introduction, to achieve both topographical and habitat restoration.
- **Marine Habitat Creation / Enhancement** - If and when habitat creation or enhancement is considered appropriate for marine aggregate sites, the **placement of shell or gravel** to encourage particular biotopes / species merits consideration, given the apparent importance of such material in the benthic ecosystem processes of natural shellfish habitats at some UK dredged sites. **Artificial reefs** may also be applicable in certain areas; regional schemes within wider management plans aimed at conservation or enhancing fisheries have proved successful in boosting benthic productivity in degraded habitats.

6.0 Assessing the Need for Remediation at Marine Aggregate Sites: Practical and Strategic Implications

6.1 Introduction

This section considers the need for remediation both in terms of government policy and the practical implications of interpreting policy. The limitations of generalised habitat data classifications in assessing the need for remediation is briefly considered, and the case for using site-specific criteria reviewed. The next two sections seek to address the question: where is remediation appropriate; i.e. for which habitats in response to which dredging conditions and judged by what criteria?

6.2 The Need for Remediation – Where is Intervention Appropriate?

In attempting to answer this question, it is prudent to begin by considering government policy, since it is the case that policy on the aftercare of sites will determine what, if any, remediation is undertaken. MMG1 makes provision for corrective action where monitoring of dredging activity identifies unacceptable impacts on the marine environment. Similarly, where monitoring indicates that the marine environment outside the dredging area is being unacceptably affected as a direct result of the dredging activity, the Secretary of State will consider carefully what action is needed to minimise further damage or, if considered necessary, to restore the area.

The crux of the issue lies in determining what is unacceptable in terms of impact. Given that some degree of natural recolonisation of all communities will occur following anthropogenic disturbance, value judgements on acceptability of impact reflect (more often implicitly than explicitly) relative timescales for the recovery of communities.

A review of the research into the impacts of dredging on the benthos and its subsequent recovery (see section 3) has identified many data gaps but has shown timescales for recovery broadly within the windows shown in Table 6.1.

Habitat Type	Typical
Shallow water mobile sands	6 months – 2-4 years
Shallow water stable gravel with transient sand	Up to circa 5 years
Stable gravels	5 - >10 years

Table 6.1: Typical recovery windows for general sand / gravel habitats following gravel extraction

The variability in impacts and recovery times associated with aggregate dredging illustrates the difficulty with attempting to set threshold values for the recovery rates of habitat types. This dilemma has been addressed in detail in Newell's review on dredging impacts and recovery.

"Despite the work that has been carried out over the past 30 years, the non-biologist could be forgiven for being bewildered by the diversity of the results and the difficulties of making more than the most general predictions on the effects of dredging activities including the extraction of minerals on biological resources" (Newell et al 1998).

Nevertheless some palpable but fundamental conclusions can be drawn from Table 6.1 and the detailed review presented in section 3.0, namely:

- **The recovery rate following dredging is related to habitat type, broadly classified by sediment composition and stability;**
- **Dredging intensity is often related to recovery period, although the impact is less pronounced in sands characterised by high disturbance than in more stable gravels.**
- **The considerable variation in recovery within and between habitat types indicates the significance of other environmental factors in recovery rates.**

A range of other interrelated environmental conditions have to be taken into consideration in assessing recovery rates including:

- Compaction and stability of pre-dredged deposits;
- Composition of pre-existing community;
- Change to sediment composition post dredging;
- Degree of defaunation;
- Degree of natural disturbance post extraction;
- Degree of anthropogenic disturbance post extraction;
- Hydrodynamic regime;
- Intensity and length of extraction activity, type of dredging, degree of screening etc.

At this stage, it is clear that some initial conclusions about the types of habitats and dredging activities that might benefit most from some active remediation process can be drawn. Clearly, stable gravel areas supporting diverse faunal communities that are subject to intensive dredging are more likely to benefit from such action than dredging in highly mobile, low diversity sands.

However, the variables that exist in terms of the seabed environment and the dredging operation dictate that meaningful assessments of recovery and the need for remediation be undertaken on a site specific basis, incorporating the local environmental factors which do not feature in less discriminate high level assessments.

6.3 Why Undertake Remediation? Facing the Hard Questions

Remediation, by definition, is based upon the premise that intervention will accelerate the natural recovery process of a habitat following anthropogenic disturbance. While this study has sought to provide evidence to support this premise, there are some fundamental questions which remain unanswered:

- **What is the definition of acceptable faunal recovery?**
- **What criteria can be used to best monitor faunal recovery?**
- **What constitutes an acceptable period for naturally occurring recovery?**

6.3.1 What is Acceptable Faunal Recovery?

Recovery has been defined earlier as:

***'The establishment of a community that is similar in species composition, population density and biomass to that previously present or at non-impacted sites**.'* (C-CORE 1996 as cited by Newell 2002)

As a workable tool for implementation of policy, this definition in itself is inadequate. To be universally and consistently applied, industry wide, it requires amplification.

Given the difficulties in applying generic thresholds for recovery of broad habitat types, an approach is needed which formalises and standardises assessment of impact and recovery across all sites while assimilating the site-specific environmental factors which control these processes.

The complexity of incorporating site-specific factors in assessing dredging impacts have been acknowledged in relation to fisheries in recent work undertaken by CEFAS. *"A key problem when comparing different dredge licence applications is that they all differ in the scale of their impact, and in the importance of the biological resources at the site. Clearly the different combinations of these features in each new dredge licence area prevents any simple comparison."* (Carlin and Rogers 2002)

CEFAS have developed and are currently using a formalised risk assessment process which standardises the data collated for environmental impact assessments and evaluates the impact of each aspect of a dredging operation on the full range of generic fisheries issues. The approach has the merits of achieving greater standardization and objectivity in data collation while incorporating informed scientific judgement in the assessment process. A description of the process and example is provided at Annex G. It is suggested that a similar model might be used to assess the impacts of dredging on benthic communities. The vulnerability of criteria such as the degree of defaunation or loss of rare / conservation species could be assessed in relation to dredging impacts allowing the risk posed to recovery to be evaluated.

However, whilst this process might allow some standardisation of the assessment of acceptable impacts, it does not necessarily address the core issue associated, namely a definitive objective for the recovery process. Is 80% of the original species an acceptable end point? Is it necessary to achieve 100% of pre-dredging species, abundance or biomass? Is ecosystem productivity rather than precise species composition a better measure?

6.3.2 What Parameters can be used to Describe Recovery?

The use of formalised risk assessment, while a valuable tool to predict threats to recovery, is not a definitive measure for describing recovery as has been noted above. Restated, one of our guiding principles is:

The performance of remediation needs to be measured.

If policy is to be consistently applied in relation to remediation there has to be universal acceptance of the criteria used to describe recovery. Most useful for regulators and industry alike, is a simple agreement on an acceptable end-point for each licence being considered. However, biological systems rarely follow such simple paths so that the identification of a fair, robust and universally applicable measure of recovery □ a defined end-point for recovery - presents one of the most significant challenges for managing marine processes.

One approach which is being attempted is the development of ecological quality objectives (EcoQOs) which have been defined as an overall expression of the structure and function of aquatic systems (Elliott 2001). They are linked to numerical standards which require the normal situation to be defined together with the limits of variability from the norm.

An example applied by the UK Group Co-ordinating Sea Disposal Monitoring for testing the changes in benthos at sewage sludge disposal sites centred on changes in primary and derived community parameters for possible impact areas and reference sites. The initial changes against baseline values for the primary variables used were: abundance (± 50%), taxa (± 20%), and biomass (± 20%) and for the derived variables H' (Shannon-Weiner information statistic, ± 20%), A/T (abundance ratio ± 50%) and B/A (biomass ratio ± 50%).

Whilst these measures may not be directly applicable to marine aggregate dredging, they do provide an insight into one possible approach. A variety of other possible EcoQOs that might be applied to marine dredged sites have recently been reviewed by CEFAS at a workshop event (Rees, 2001 and Boyd and Rees 2001).

It is clear that caution needs to be exercised with the use of EcoQOs and introduction of EcoQS particularly in relation to:

- The sampling strategy and statistical methods employed (Rees 2001);
- The lack of baseline environmental data for some older dredging sites; (Boyd and Rees 2001
- The paucity of relevant impact studies and time-series data for coarse ground habitats (Rees 2001).

Accepting the above shortcomings, the method of parallel sampling at treatment (aggregate) sites and reference sites, with rigorous quality assurance, would appear to offer a plausible means of measuring change from a baseline condition. In this respect it could be used to monitor the process of natural recovery, following dredging. Where natural recovery falls significantly behind expectations (derived from a site-specific risk assessment) it could serve as a trigger for appropriate intervention. The success or otherwise of remediation in aiding recovery could then be judged by the same criteria.

Use of EcoQOs in this context would be solely on a site-specific basis and would not link to any wider threshold values which await the collation and analysis of more comprehensive data on the impacts of dredging on recovery rates.

Figures 5.6a and 5.6b are a graphical representation of how monitoring against site specific EcoQOs would be used as a trigger mechanism to manage dredging impacts through mitigation or specification of remediation requirements.

The shading within the flow diagram represents ecosystem health. Light / greens browns represent background conditions in response to varying degrees of natural stress. These are higher in figure 5.6a for a high stress mobile sand environment than in figure 5.6 b where natural conditions are more stable. Stress on the environment introduced by dredging (red) results in perturbed conditions shown by the darker browns.

What is lacking in the present regulatory structure is the trigger mechanism depicted by the minimal / moderate and unacceptable thresholds of impact. It is these which would be based upon EcoQOs determined on a site-specific basis, both from reference site conditions and, where available, from baseline conditions prior to commencement of dredging.

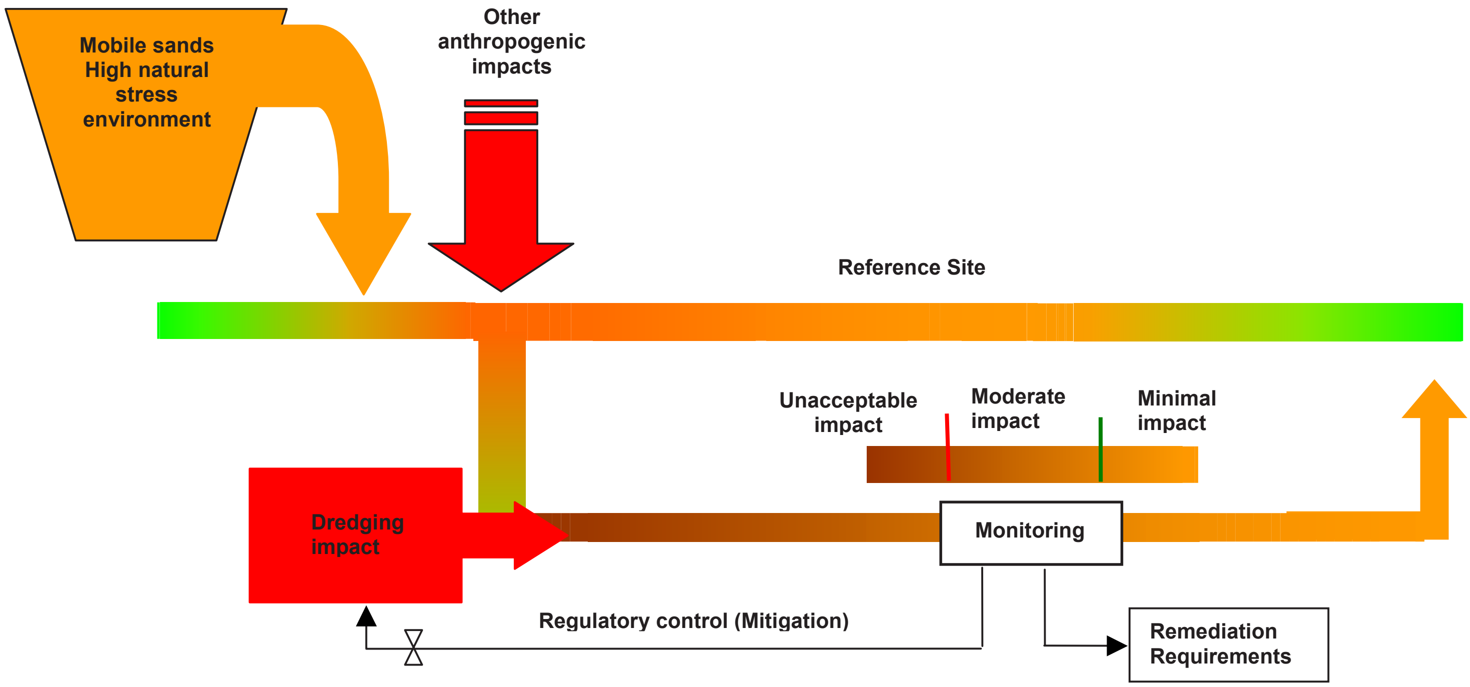


Figure 5.6a Conceptual diagram of monitoring and trigger mechanism for a mobile sands aggregate site.

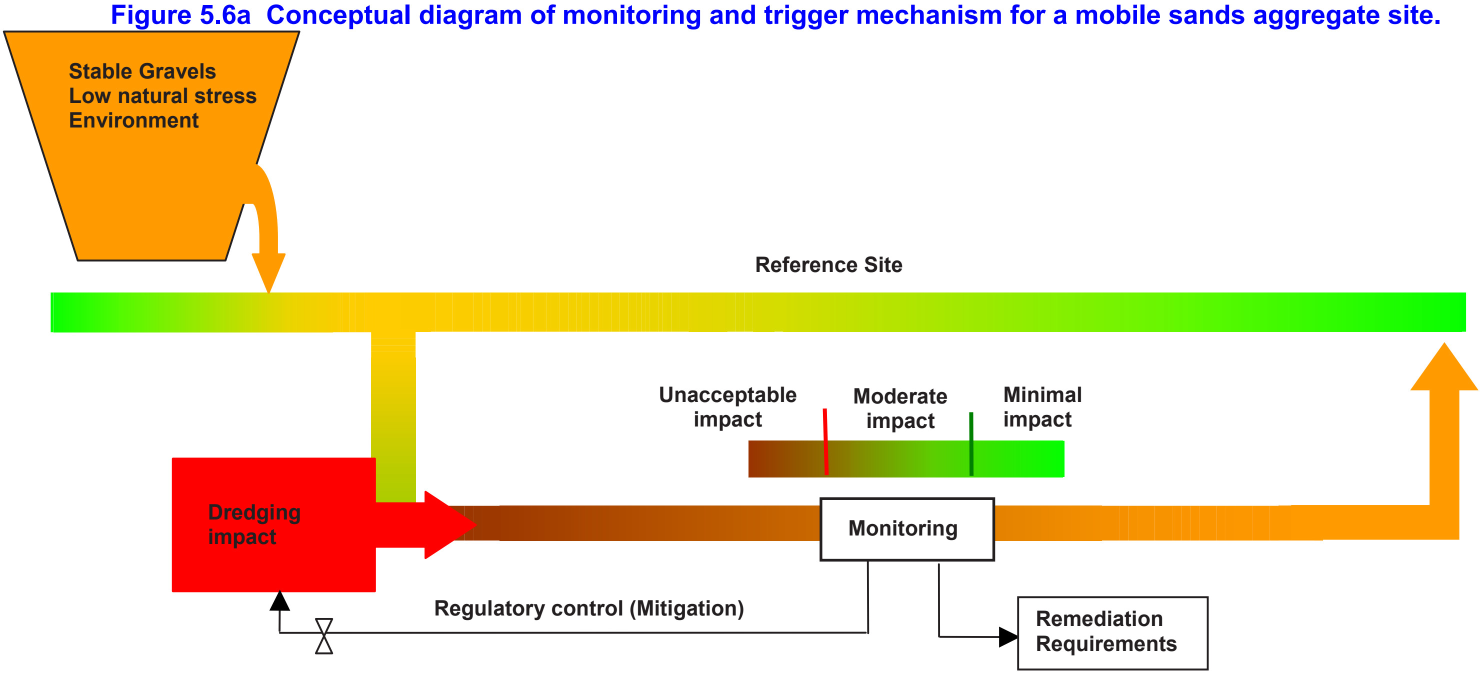


Figure 5.6b Conceptual diagram of monitoring and trigger mechanism for a stable gravel aggregate site.

Where perturbation from background conditions is considered to be unacceptably high, this would trigger appropriate mitigation (e.g. a reduction in the intensity of dredging) and / or specification of remediation requirements.

Key points to note from the diagrams are:

- The determination of unacceptable / acceptable thresholds of impact is different for the two sites based upon site specific conditions. Equally, these could be quite different for two broadly similar habitat types due to local environmental conditions.
- The determination of background levels of ecosystem health is further complicated by the perturbation introduced by other anthropogenic stresses e.g. from diffuse pollution or trawling activity for which the impact is very difficult to quantify.
- While the diagrams are intended to represent mechanisms for active sites, a similar approach and trigger mechanism could be put in place to initiate remediation at a site where extraction has ceased. In this instance the trigger would be based upon comparison of expected vs. actual timescales for recovery.

6.3.3 What Constitutes an Acceptable Period of Natural Recovery?

It might be reasonable to suggest that dredging in naturally disturbed sands, for which a site-specific assessment has predicted recovery of the fauna post-dredging within 4 to 6 months, is wholly inappropriate for remediation and recovery should proceed naturally. At the other end of the spectrum, there is arguably a stronger case for remediation, post dredging, if it can be demonstrated that the natural recovery period for a stable gravel site characterised by a rich and diverse faunal community could be reduced from say 15 to 5 years. Within these two extremes however, there is a need to develop a clear, sound, scientifically based, and economically viable criteria for assessing where remediation may or may not be appropriate.

Dredging clearly has impacts on the environment but remediation can also have impacts and involves the commitment of resources and money. Furthermore, it is a relatively untried process in the marine environment suggesting complex management issues will need to be addressed. The costs of remediation which are ultimately borne by society as a whole have to be weighed up against the ecological benefits which they obtain. This process requires an objective measure of ecological value.

It is suggested that any attempt to apply absolute monetary values to ecological assets in this context is meaningless. Rather, the costs and benefits of remediation options need to be considered in relative terms, one to another, and perhaps relative to the alternative benefits which could be obtained from the monies involved.

We can illustrate this point with a simple hypothetical example.

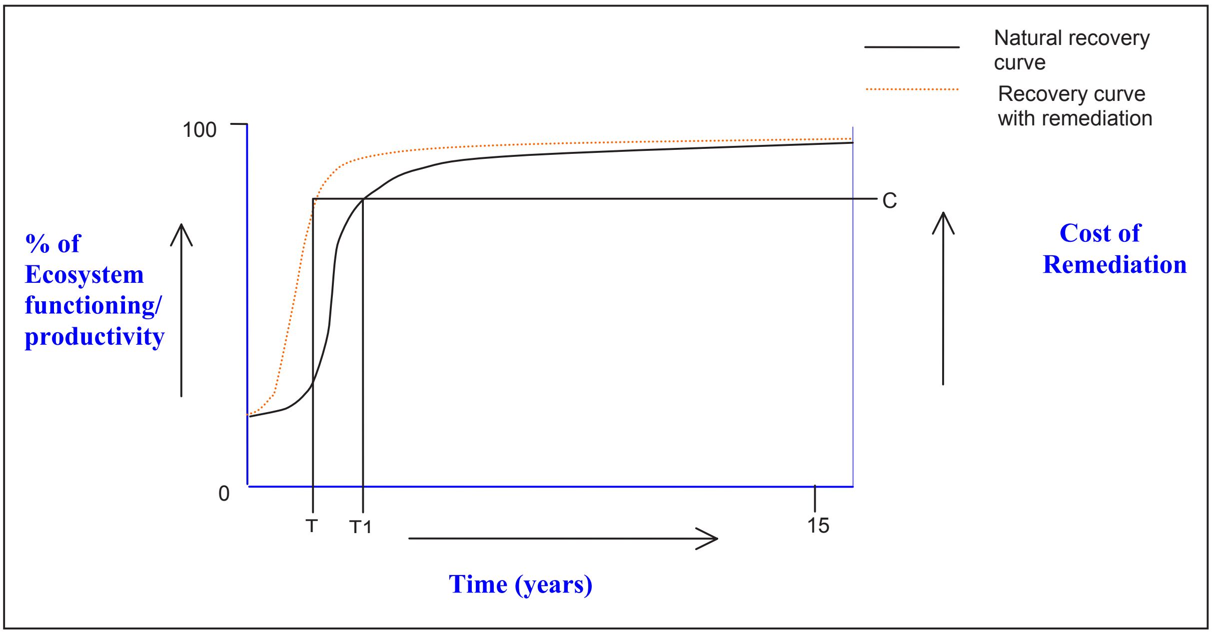


Figure 6.1: Hypothetical recovery curves for shallow water mobile sands

Figures 6.1, 6.2 and 6.3 depict recovery curves, post dredging, for shallow water mobile sands, a shallow water gravel habitat with transient sand and a stable gravel habitat respectively. The red curve for each graph represents the theoretical rate of recovery which could be achieved by means of imposing a remediation policy at the site at cost C. The benefits of the remediation for the three habitats can be viewed in terms of the relative effects on recovery rates. All other considerations aside, the same cost would achieve a marginal benefit in figure 6a (T1-T is small) a somewhat higher benefit in figure 6.2 and a potentially significant benefit in figure 6.3 (T1-T is large). One can also consider the relative sizes of the areas between the curves in each example which represent the marginal gains in terms ecosystem functioning and productivity which will be obtained from the remediation applied.

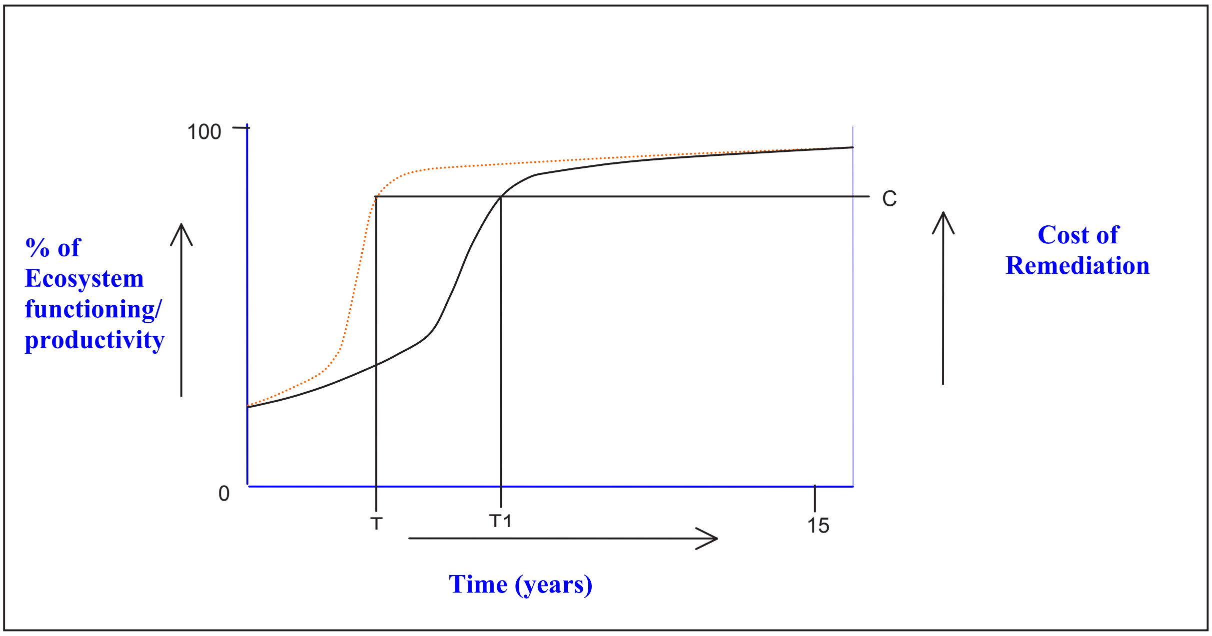


Figure 6.2: Hypothetical recovery curves for stable gravel with transient sand

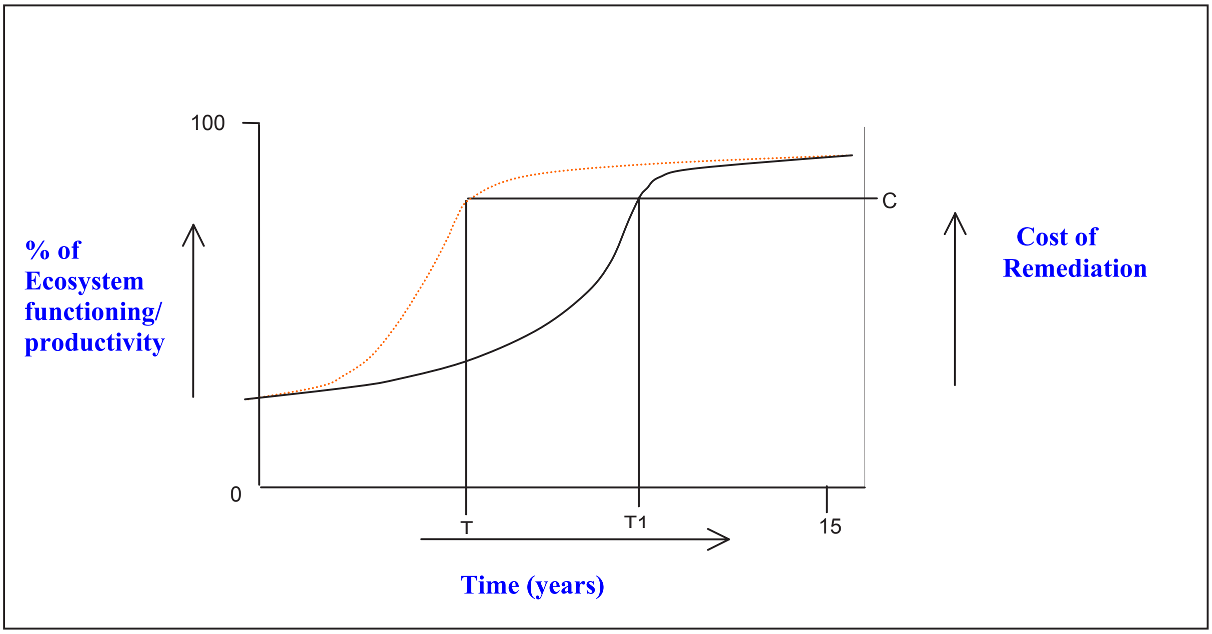


Figure 6.3: Hypothetical recovery curves for stable gravel with transient sand

These graphs are of academic interest only, until one is able to enter actual, derived values. While it would be relatively easy to estimate the costs, a fairly rigorous series of field trials would need to be undertaken to ascertain the relative rates of recovery for the three habitat types and the effects which a given remediation approach would have on these recovery rates. Furthermore, as has been noted above, there would be a need to assess the environmental impacts of any remediation against the ecological benefits that might accrue and the economic cost of that action.

6.4 What Methods of Remediation have an Application to Former Dredging Sites?

By considering the impacts of dredging on benthic recovery (section 3) and the methods of remediation (section 5) a summary of potential remediation methods has been compiled (Table 6.2) where appropriate suggesting potential applications. The potential benefits and shortcomings listed for each option would form the starting point for a detailed cost benefit analysis for a given project. Several points emerge from this exercise namely:

- **The spatial extent of impact is a limiting constraint on the appropriateness of a number of remediation methods.**
- **Active-Passive remediation is a flexible tool which, if proven, could be applied over wide areas.**
- **Conversely physical restoration / enhancement methods including gravel seeding / profiling may only be appropriate to individual features / or targeted areas. These probably merit consideration as part of wider marine management / biodiversity plans.**
- **Artificial reefs may form a useful part of any remediation scheme and act as compensation in terms of productivity/diversity on a regional basis.**

It is emphasised that Table 6.2 is in no way intended to be prescriptive but has been compiled to stimulate the debate on remediation in the marine environment from which it is hoped a consensus will emerge as to where research priorities should be focused for trialling some of these techniques within the field.

Remediation method	Potential Benefits	Potential Shortcomings	Potential Applications?
Non Intervention The method currently employed at most existing marine aggregate sites in conjunction with monitoring programs.	• Inexpensive. • No secondary impacts from remediation. • Relies upon natural processes.	• No control. • Slower recovery than with intervention? • Limited applicability? • No additional benefits to biodiversity etc. • No control of cumulative impacts. • Continuing other damaging uses / activities.	• Habitats characterized by high rates of natural disturbance subject to rapid recovery, areas characterised by relatively 'low value' faunal communities or low dredging intensity sites.
Active-Passive Recovery Relies upon natural recovery with the additional measure of designating a non-disturbance zone / Marine Protected Area (MPA)	• Allows recovery by natural processes. • Flexible □can be moulded to management objectives. • Can be used to remove / minimise cumulative impacts. • Necessitates stakeholder engagement. • Universally applicable - i.e. could apply to whole sites (requires qualification). • Potentially higher fishing yields adjacent to non-disturbance zones.	• Secondary impacts on other sea users. • Expensive □could involve compensation to fishing industry. • Administratively complex □legality and licensing issues. • Enforcement issues. • Could transfer / concentrate impacts in other areas.	• Areas in which cumulative impacts from other activities are considered to be significant. • Specific areas within dredging sites which are found to be ecologically impoverished. • Research application to assess the benefit of this option as a remediation measure which is currently unknown.
Preferential Use of former Aggregate Sites Planning measures which take into consideration the degraded condition of a site	• Would allow the construction of e.g. wind farms to occur in impoverished areas. • Wind Turbine construction could provide a relatively undisturbed seabed environment in the longer term to allow recovery.	• Would result in further disturbance in the short term. • A 'brownfield' mentality could develop where the primary focus of former extraction sites is no longer recovery.	• Faunally impoverished sites.

Table 6.2: Summary of remediation methods, potential benefits / shortcomings and potential applications

Remediation method	Potential Benefits	Potential Shortcomings	Potential Applications?
Conserving the Modified Seabed Conservation of former dredging sites deemed to have intrinsic conservation value.	• Biodiversity gains? • Low cost? • Necessitates stakeholder engagement.	• Loss of pre-dredged habitat. • Limited applicability. • Management and cost implications? • Impact on other users □trawlers.	• Examples found to date, relate to static depressions.

Physical Restoration / Profiling
 	
Variety of methods including filling static depressions, seeding of scallop shells to restore the physical nature of the seabed pre-dredging thereby encouraging faunal recovery.
	• Biodiversity gain?
	• Enhanced recovery rate?
	• Positive impact for fisheries
	• Beneficial use of waste materials?
	  ○ Processed scallop shells?
	  ○ Rejected aggregate dredgings?
	  ○ Maintenance dredgings?
	SCALE DEPENDANT
		• High cost?
		• Not feasible for large areas?
		• Technologically unproven?
		• Legality and licensing issues.
		• Secondary impacts
			• Research applications to assess the benefits of these options as remediation measures, which are currently unknown
			• Filling / capping of carefully selected static depressions considered to be ecologically impoverished?
			• Highly targeted capping of specific areas where sediment composition has changed radically due to the intensive dredging operations?.

Habitat Creation / Enhancement - Reefs	• Increased biodiversity? • Increased productivity? • Secondary benefits for fisheries? • Amenity uses □divers? • Necessitates stakeholder engagement. • Could be linked with regional biodiversity targets. • "Feel good" factor. • Beneficial use of waste? **SCALE DEPENDANT**	• Expensive. • Physical obstruction for other sea users / trawlers • Artificial intervention in existing ecosystems. • Uncertain benefits. • Hydrodynamic impacts. • Not universally applicable □ small scale applications only.	• Targeted use of habitat enhancement measures at suitable sites may be viable to increase productivity and/or biodiversity of an area/region. • Habitat enhancement may have potential as a component of a regional management plan.

Table 6.2(contd.) Summary of remediation methods, potential benefits / shortcomings and potential applications

6.5 Charting the Way Forward

In determining the place for remediation in the marine aggregate industry, the role of the regulator is pivotal. The licensing process places exacting conditions on operators in relation to their activities to minimise the impacts of dredging on the seabed (see section 2.5.2). Environmental data from aggregate sites are monitored by the government in conjunction with dredging activity data in order to review impacts and intervene where necessary.

The detailed requirements embodied in the government's precautionary approach to licensing and monitoring should form the basis for a thorough consideration of remediation, taking into account all of the issues associated with this potentially complex process. **Currently, the powers of the Secretary of State to intervene where he or she considers that restoration objectives post-dredging will not be met are not fully executed because there is no objective, agreed, effective trigger mechanism in place at most sites.**

Development of an appropriate mechanism for restorative action is considered an essential step in remediation policy for marine aggregate sites and one that can only be achieved through broad consensus of all parties concerned. Such a system should guard against indiscriminate regulatory measures governing mitigation that could be cost-prohibitive for developers or non-beneficial for the environment. The critical role of government, therefore, is to balance the reasonable and economically viable (private) project related costs with the current and future potential costs to the marine environment and other sea users associated with a minimalist approach to mitigation. A debate on the criteria for intervention will help to ensure that suitable mitigation, specifically in the form of remediation, is applied in a targeted and appropriate, site-specific manner and thereby ensure an adequate response to post-dredging site management issues.

A further point arises, i.e. the current plans for the future designation of offshore areas as possible SACs or SPAs as part of the Natura 2000 network and the possible role for the marine dredging industry in assessing, monitoring and managing such areas. Similarly BAPs or SAPs may benefit from an agreed process of site management much as they have in the terrestrial environment.

In seeking to progress the development of appropriate policies and a variety of techniques that should be applied to the management of licensed areas following the cessation of dredging, there is clearly a significant amount of further assessment required. We consider this task might best be taken forward by a Marine Dredge Site Management Working Group composed of industry representatives, regulators, key stakeholders and technical experts. The recommendations provide some key issues for them to consider in their plenary session.

6.6 Recommendations

- **Establish a marine dredged site management working group involving industry, regulators, key stakeholders and technical experts.**

- **Consider priorities for further targeted research into dredging impact and recovery rates, specifically related to a variety of dredging intensities in the most sensitive habitats (i.e. stable gravels).**

- **Develop robust criteria for assessing recovery and determining when intervention in the form of remediation is necessary.**

- **Explore the contribution which the aggregates industry can make to marine BAPs and SAPs and to participation in site management of future offshore Natura 2000 sites.**

Bibliography

ABP, 1998. Review of Coastal Habitat Creation, Restoration and Recharge Schemes. Report Number R.909, ABP Research and Consultancy Ltd 1998.

Atkinson P, Crooks, S, Grant A, Rehfisch M, 2001. The success of creation and restoration schemes in producing intertidal habitat suitable for waterbirds. English Nature Research Report No. 425, July 2001.

BMAPA / The Crown Estate, 2004. The Area Involved, 5th Annual Report (2002), 2004.

Booth J., 2000. Marine enhancement in New Zealand potential species and areas. A report prepared for Ministry for the environment. National Institute of Water and Atmospheric Research Ltd November 2000.

Boyd, S.E. and Rees, HL., 2001. Towards an Understanding of the Marine Aggregate Extraction on the Seabed as a Basis for Developing an ECoQOS Approach for Management. [Internet] available at: http://www.cefas.co.uk/news/news020801a.htm

Boyd, S. E., and Rees, H. L., 2003b. An examination of the spatial scale of impact on the marine benthos arising from marine aggregate extraction in the central English Channel. Estuarine, Coastal and Shelf Science, 57; 1-16.

Boyd, S. E., Limpenny, D. S., Rees, H. L., Cooper, K. M. and Campbell, S., 2003a. Preliminary observations of the effects of dredging intensity on the re-colonisation of dredged sediments off the southeast coast of England (Area 222). Estuarine, Coastal and Shelf Science, 57; 209-223.

Brown, C.J., Hewer, A.J., Meadows, W.J., Limpenny, D.S., Cooper, K.M., Rees, H.L. and Vivian, C.M.G., 2001. Mapping of gravel biotopes and examination of the factors controlling the distribution, type and diversity of their biological communities. CEFAS Science Series Technical Report No. 114. (DFRA Research Project AE0908) Lowestoft.

Ceccaldi H., 2002. Artificial Reefs versus Underwater Structures to Enhance Ecology and Fisheries: An attempt of classification and future ways of research. Biologia Marina Mediterranea Vol. 9 (2) p43-50 2002

Cefas (2001) Workshop on Ecological Quality Objectives for Marine Aggregate Extraction Presentations [Internet] available at: http://www.cefas.co.uk/news/eco-programme.pdf

Chisholm S., 2000. Stirring times in the Southern Ocean. Nature 407 (6805) p685-687.

Cripps S., Picken G., Aabel J.p., Andersen O.K., Heyworth C., Jakobsen M., Kristiansen R., Marken C., Paulsen J.E., Shaw D., Annand A., Jacobsen T.G. and Henriksen I.B., 1998. Disposal of oil-based cuttings Report RF-98/097 for The Norwegian Oil Industry Association. Rogaland Research April 1998.

De Groot S.J., 1986. Marine Sand and Gravel Extraction in the North Atlantic and its Potential Environmental Impact, with Emphasis on the North Sea. Ocean Management, 10: 21-36.

De Groot, S.J. 1979. An Assessment of the Potential Environmental Impact of Large Scale Sand Dredging for the Building of Artificial Islands in the North Sea. Ocean Management, 5: 211-232.

DECODE, 2001. Beneficial Use of Fine-Grained Maintenance Dredged Material, **D**etermination of the **E**cological **C**onsequences of **D**redged-material **E**mplacement (DECODE) - www.cefas.co.uk/decode/use

Desprez, M., 2000. Physical and biological impact of marine aggregate extraction along the French coast of the Eastern English Channel. Short and long-term post-dredging restoration. ICES Journal of Marine Science 57, 1428-1438.

Doody J.P., 2003. Coastal Habitat Restoration-towards good practice, Introductory Report. Part of the LIFE funded Living with the Sea Project. National Coastal Consultants.

Doody J.P and Randall R.E., 2003. A guide to the Management and Restoration of Coastal Vegetated Shingle. Contract No. MAR 05-03-002 English Nature

Duncan M & Stark K., 2000. Elliot Bay/Duwamish Restoration Project, Elliot Bay Nearshore Substrate Enhancement Monitoring Plan. Elliot Bay/Duwamish Panel Publication No 25. May 2000.

Dutch Ministry of Transport and Public Works, 1991. Regional Extraction Plan for the Dutch Part of the North Sea. Environmental Impact Statement. (RON/EIS). October 1991.

Emu, 2000. Area 436/202 Cross Sands Benthic Survey. Report No. 00/J/1/03/0145/0179.

Emu, 2002a. Area 254 Cross Sands Benthic Ecology Study. Report No. 02/J/1/03/0408/0300.

Emu Ltd., 2002b. Hastings Shingle Bank. Macrobenthic Ecology Study (2001). Report No. 02/J/1/03/0339/0323, to the Resource Management Association.

Emu Ltd., 2002c. Inner Owers Dredge Pit Comparison. Report No. 02/J/1/03/0378/0270/Final, to Hanson Aggregates Marine Ltd.

Emu Ltd., 2002d. Nash Bank Dredging Licence Area Benthic Ecology Study. Report No 02/J/1/03/0244/0295.

Gall M, Ross A, Zeldis J and Davis J., 2000. Phytoplankton in Pelorus Sound food for mussels. Water and Atmosphere 8 (3) p8-10.

Horwood J., 2002. Fishery Exclusion Zones: not all good, not all bad (electronic presentation, www.cefas.co.uk/fsmi/exclusion_zones.pdf). CEFAS, Lowestoft.

Jennings S., 1999. The Case for No-Take Zones. Fishing News 12th March 1999.

Johnson C.M., Turnbull C.G., Tasker M.L., 2002. Natura 2000 in UK Offshore Waters: Advice to support the implementation of the EC Habitats and Birds Directives in UK offshore waters. JNCC Report 325.

Kenny A. J., and Rees H. L., 1994. The Effects of Marine Gravel Extraction on the Macrobenthos: Early Post-dredging Recolonisation. Marine Pollution Bulletin, Vol. 28, No. 7, pp. 442-447.

Kenny A. J., Rees, H. L., and Lees, R. G., 1991. An inter-regional comparison of gravel assemblages off the English east and south coasts: Preliminary Results. ICES CM 1991/E:27.

Kyriakopoulos C., Tselentis B. and Frantzis A., 2002. Underwater Marine Protected Areas: Proposing a Novel Tourist Attraction within an Effective Coastal Zone Management Plan for Greece. From first Scientific Conference on Oceanographic Aspects for a Sustainable Mediterranean, Athens 2002. European Federation of Marine Science and Technology Societies.

López-Jamar, E. & Mejuto, J. 1988. Infaunal Benthic Recolonisation after Dredging Operations in La Coruña Bay, NW Spain. Cah. Biol. Mar. 29: 37-49.

Millner, R. S., Dickson, R. R., and Rolfe, M. S., 1977. Physical and biological studies of a dredging ground off the east coast of England. ICES CM, 1977/E:48. 1-11.

Newell, R.C, Seiderer, L. J., Simpson, N. M., and Robinson, J. E., 2002. Impact of Marine Aggregate Dredging and Overboard Screening on Benthic Biological Resources in the Central North Sea: Production Licence Area 408, Coal Pit. Marine Ecological Surveys Ltd Technical Report No. ER1/4/02 to the British Marine Aggregate Producers Association. 72pp.

Newell, R.C., Seiderer, L. J., Hitchcock, D.R., 1998. The Impact of Dredging Works in Coastal Waters: A Review of the Sensitivity to Disturbance and Subsequent Recovery of Biological Resources on the Seabed. Oceanography and Marine Biology: an Annual Review 36: 127-78.

Office of the Deputy Prime Minister, 2002. Marine Minerals Guidance Note 1: Guidance on the Extraction by Dredging of Sand, Gravel and Other Minerals from the English Seabed. Consultation Paper. ODPM, London.

Palermo M., Shafer R., Brannon J., Myer T., Truitt C., Zappi M., Skogerboe J., Sturgis T., Wade R., Gunnison D., Griffin D., Tatum H. and Portzer S., 1989. Evaluation of dredged material disposal alternatives for US Navy Homeport at Everett Washington, “Technical Report EL 89-1, US Army Engineer Waterways Experiment Station, Vicksburg, Mississippi.

Peyre, Bush and Plunket, 2002. Coastal Restoration, Rebuilding Fishery Habitat. Louisiana Agriculture Vol. 45, No. 2, 2002.

Poiner, I.R. & Kennedy, R., 1984. Complex Patterns of Change in the Macrobenthos of a Large Sandbank Following Dredging. Marine Biology 78: 335-352.

Polunin N. and Wabnitz C., 2001. Marine Protected Areas in the North Sea. Briefing Paper 1, Fisheries Society of the British Isles (FSBI). Granta Information Systems 2001.

Quarry Products Association, 2003. ODPM Consultation: revision of national regional guidelines for aggregate provision in England 2001-2016.

Rees, H. L., 1987. A survey of the benthic fauna inhabiting gravel deposits off Hastings, Southern England. ICES CM, 1987/L:19, 1-19.

Rees, H.L., 2001. Practical considerations relating to the derivation of Ecological Quality Objectives for marine aggregate extraction activities [Internet] available at: http://www.cefas.co.uk/news/news020801a.htm

Russell, M 2003. Marine Aggregates: Past, Present and Future transcript of address to marine biodiversity conference.

Sakshaug E., Olsen Y. and Hoell E., 1995. Maricult research programme: Fertilising marine ecosystems to harvest more. Council meeting of the International Council for the Exploration of the Sea, Aalborg p21-29, September 1995.

Sardá, R., Pinedo, S., Gremare, A. & Taboada, S., 2000. Changes in the Dynamics of shallow Sandy-bottom Assemblages due to Sand Extraction in the Catalan Western Mediterranean Sea. ICES Journal of Marine Science, 57: 1446-1453.

Seaman W. and Jensen A., 2000. Purposes and practices of artificial reef evaluation. In: Artificial reef evaluation with application to natural marine habitats (Ed. Seaman W.) CRC Press 2000.

Shelton, R.G.J. and Rolfe, M.S., 1972. *The biological implications of aggregate extraction: recent studies in the English Channel.* Report to ICES by MAFF Fisheries Laboratory.

Sheng Y., 2000. Physical characteristics and engineering of reef sites. In: Artificial reef evaluation with application to natural marine habitats (Ed. Seaman W.) CRC Press 2000.
UK Marine SACs Project (2001) Natura 2000 European Marine Sites ecological sensitivity and management requirements.

US Army Corps of Engineers, 1999. Implementation Strategy of the Dredged material Management Plan for the Port of New York and New Jersey Technical Appendix. US Army Corps of Engineers New York District September 1999.

US Army Corps of Engineers Waterways Experiment Station, 1989. Environmental Effects of Dredging EEDP 07-1 07-5.

Van Dalfsen, J.A., Essink, K., Toxvig, H., Madsen, J., Romero, J. & Manzanera, M., 2000. Differential Response of macrozoobenthos in the North Sea and the Western Mediterranean. ICES Journal of marine Science, 57: 1439-1445.

Van der Veer, H.W., Bergman, M.J.N. & Beukema, J.J., 1985. Dredging Activities in the Dutch Wadden Sea: Effects on Macrobenthic Infauna. Netherlands Journal of Sea Research. 19: 183-190.

Van Moorsel, G. W. N. M., 1994. The Klaverbank (North Sea), geomorphology, macrobenthic ecology and the effect of gravel extraction (pp. 1-65). Bureau Waardenburg, Culemborg, Report No. NR 94.24.

Van Moorsel, G. W. N. M, and Waardenburg, H. W., 1991. Short-term recovery of geomorphology and macrobenthos of the Klaverbank (North Sea) after gravel extraction (pp. 5-54). Bureau Waardenburg, Culemborg, Report No. NR 91.20.

Wardell Armstrong 1996. Reclamation of Damaged Land for Nature Conservation. (HMSO)

White, G. J. and Gilbert, J. C. (eds), 2003. Habitat creation handbook for the minerals industry. The RSPB, Sandy.

Glossary

Amphipod	small crustacean with a flattened body (e.g. sand flea).
Benthic	of the seabed.
Biogenic reefs	reefs created by living organisms (e.g. ross worm).
Biomass	the total quantity of living organisms in a given area, expressed in terms of living or dry weight, or energy value per unit area.
Biotope	the physical habitat with its associated biological community; this term refers to the combination of a physical environment and its distinctive assemblage of conspicuous species.
Bivalve mollusc	with two hinged shells (e.g. mussel, cockle)
Bryozoan	member of the Phylum Bryozoa (‘moss animals’). Colonial, filter-feeding invertebrates that form elaborate, branched or encrusting colonies attached to a hard substrate.
Circalittoral	subtidal zone dominated by animals, where low light levels limit algal growth.
Echinoderm	member of the Phylum Echinodermata; invertebrate animal with 5-rayed body symmetry e.g. starfish, brittlestar, sea urchin.
Epifauna	animals living attached to the surface of the substrate.
Hydroid	member of the Phylum Cnidaria, Class Hydrozoa. Colonial animal forming branched or unbranched colonies attached to the substratum.
Mysidia	Shrimp like creatures less than 2.5 cm long, recognisable by their appendages; they have no nippers but eight pairs of branched swimming legs, enclosed by a thin carapace.
Infralittoral	subtidal zone dominated by plants.
Infauna	animals living beneath the surface of the substrate.
Polychaete	worm of the Phylum Annelida, Class Polychaeta possessing many segments with paired appendages (e.g. ragworm).
Sessile	fixed in one position; immobile
Sublittoral	below the low water mark
Tunicates	also known as sea squirts are small filter feeding invertebrates of the Phylum Chordata.

ANNEX A

Table of Dredging Impacts on Benthic Habitats

	Habitat and Community		
Static Dredging	**Shallow water mobile sand**	**Shallow water stable gravel with transient sand**	**Shallow/Deep water stable gravel**
Spatially localised High intensity Low frequency	(van Der Veer *et al*, 1985). Dredging for dyke construction resulted in the creation of pits 10m deep with a bottom devoid of fauna. (Dutch Ministry of Transport & Public Works, 1991) An Environmental Impact Assessment predicted the destruction of bottom fauna following dredging in sand with rapid recovery by opportunist species, via larval migration, over a period of 2-3 years.	No data	Shelton and Rolfe, 1972. Examination of static/anchor dredge pits, previously dredged two years or more before. Colonisation by muddy sand habitat species with additional gravel based fauna occupying due to sediment subsidence. Sand accumulating and consolidating in the bottom of the hole. Recolonisation well advanced, by a mixture of sand and gravel fauna. (Newell *et al*, 1998). Dredge pits/furrows in depths of +30m may persist due to the reduced or lack of wave induced sediment mobility.
Spatially localised High intensity Medium frequency	In low dynamic Mediterranean environments (Costa Dorada) where the maximum extraction depth was 2.0m, a large change to the sediment structure occurred following dredging. No faunal detail given.		
Spatially localised High intensity High frequency	No data	Boyd, *et al,* 2003. Area 122/3 east of Bembridge, IOW. Static Dredging. Sediments across whole study area were heterogeneous though significantly higher gravel contents were recorded within the intensive dredge area. Species suppressed at centre of dredging activity compared with site 500m from activity, though lowest values were recorded at 2000m. High abundance was recorded at distances of 500 and 1000m from dredging, particularly as a result of the high numbers of filter feeders, *Dendrodoa grossularia, Balanus crenatus, P. lamarcki* and *C. fornicata*. The centre of the dredging was characterised by a diverse but variable community (from k-dominance curves), though 2000m site (furthest site from dredging) was identified as having highest diversity (even with lowest total number of species). Site at 1000m exhibited highest dominance.	Shelton and Rolfe, 1972. Examination of static/anchor dredge pits, previously dredged within one year. Almost completely defaunated (2 spp and 4 individuals). Colonisation by muddy sand habitat species, with muddy sand accumulating in the bottom of the hole.

Habitat and Community			
Trailer	**Shallow water mobile sand**	**Shallow water stable gravel with transient sand**	**Shallow/Deep water stable gravel**
Restricted area Low intensity Low frequency	(de Groot, 1979). Dredging for sand in the Gulf of Mexico (Alabama). No detail on level of effect at immediate impact site, although faunal enhancement was noticed especially in adjacent depositing areas.	(van Moorsel & Waardenburg, 1991, van Moorsel, 1994) Studies on the Klaverbank (Cleaver Bank, N. Sea). Reduction in diversity (30%), abundance (72%) and biomass (80%). 20% of seabed in experimental area was dredged over 2 month period. Opportunistic sand-dwelling fauna invaded sand patches within the area in furrows between gravel waves.	No data
Restricted area Low intensity High frequency	(ICES, 1979). A side scan sonar survey of dredged coarse sand (350μm median dia.) deposits indicated the presence of trailer tracks but no information on faunal impacts. (de Groot, 1979). Oregon (Pacific) dredging for 8000m3 sand. No detail on level of impact although noted that fauna within the adjacent area affected by deposition of fine material. (Lopez-Jamar & Mejuto, 1988). Random dredging events (approx. days apart) within a harbour area over 6 months, in a mud and slightly sandy mud sediment at La Coruña Bay. During dredging, diversity was reduced to a few species.		
Restricted area High intensity Medium frequency			Interpolated from high frequency - Expect similar species composition although slightly reduced number of taxa, partial recovery of both biomass and abundance. Expect majority of abundance in only a few species.
Restricted area High intensity High frequency			Kenny et al 1998. Experimental study, impacts immediately after dredging were 65-70% reduction in taxa, >90% reduction in abundance, >90% reduction in biomass. Species composition broadly similar although reduced rare species. Dominants same. Transient sand/disturbance species e.g. *Dendrodoa* and *Balanus* initial species after dredging.
Large area Low intensity High frequency		Boyd et al, 2003 Area 351 east of Bembridge, IOW. Trailer suction dredging. Within extraction site, diversity richness and species number depressed compared with stations at distance from works (2000m and 5000m). Diversity and richness also reduced at 1000m. Significant effect evident from within area of dredging to 1000m distance with samples dominated by few species within dredge area and greater diversity evident away from dredging activities. Sediments were heterogeneous across the whole survey area though sites at 2000m did have higher fines content. Dredging intensity was found to correlate significantly with species composition but was not strongly linked with sediment variables.	HSB (Emu, 2002) generally low intensity but continuous dredging over four years, resulting in reduced diversity (by 70%), abundance (by 55%) and biomass (by 80%). Little or no epifauna. Dominant species *Poecilochaetus serpens* and *Scalibregma inflatum*.

Habitat and Community			
Trailer	**Shallow water mobile sand**	**Shallow water stable gravel with transient sand**	**Shallow/Deep water stable gravel**
Large area High Intensity Low frequency			HSB (Emu , 2002) Immediate impact area not affected for over 5 years, faunal community fully recovered with high diversity, abundance and biomass. Balanced epifaunal and infaunal assemblages with key species indicative of stability. Outer Owers (Emu, 2003). Immediate impact area not dredged for 8+ years. No data on impacts at the time but adjacent experimental dredging indicated substantial loss of fauna at the base of the dredge tracks, with significantly affected areas on the edges of the tracks, but relatively unaffected areas further from the tracks (10m).
Large area High Intensity Medium frequency			HSB (Emu, 2002) spatially extended area and high intensity dredging but reduced annual tonnage. Reduced diversity (by 45%), abundance (by 30%) and biomass (by 50%). Mixture of infauna and epifauna although most frequently present species were polychaetes.
Large area High Intensity High frequency	(Sardá *et al*, 2000). 150ha coarse-medium sand suction dredged between summer November 1994 in Catalonia (Mediterranean). Almost complete defaunation occurred immediately after dredging. (Poiner & Kennedy, 1984). 14M cubic metres of fine sand removed over two years from a 5km^2 section of a large subtropical sandbank at Moreton Bay, Queensland. There was a 50% reduction in diversity and a 60% reduction in abundance within the dredged area. Over adjacent areas (up to 2km from the dredge site), diversity and abundance increased significantly. Benthic enhancement was attributed to the release of resources. There was little change in the sediments during dredging.	Newell *et al,* 2002. Area 408 Coal Pit. Two areas dredged heavily , one of which was dredged heavily up to 24 months prior to survey and then only lightly until 12 months before survey; the other site was heavily dredged until 36 months and lightly until 12 months prior to survey in 2000. Dredged area was worked immediately prior to survey. Active dredge zone showed no significant suppression of species or abundance, the whole area being characterised by opportunists. Greater dominance at dredged stations within 500m of activity. Biomass within actively dredged zone was suppressed in comparison with surrounding deposits (82% reduction). Active area showed marked increase in the contribution of polychaetes to biomass with major decline in mollusc, crustacean and echinoderm species. Body size was also depressed in comparison with surrounding areas indicating re-colonisation process ongoing even while dredging is being conducted within the area. Dredge trails rapidly in-filled by transient sands.	HSB- (Emu, 2002) Immediate impact area = reduced diversity (by 70%), abundance (by 55%) and biomass (by 80%). Little or no epifauna, dominant species *Poecilochaetus serpens* and *Scalibregma inflatum.*

Habitat and Community			
Trailer	**Shallow water mobile sand**	**Shallow water stable gravel with transient sand**	**Shallow/Deep water stable gravel**
Large area High Intensity High frequency (cont.)		Desprez, 2000 Offshore Dieppe. Maximum impacts: 80% reduction in species, 90% reduction in abundance and 83% reduction in biomass. Overall effect from dredging, drawn from 1993 survey data, was 63% reduction in species, 86% reduction in abundance and 83% reduction in biomass. Community structure changed from coarse sands (and gravel) with *Branchiostoma lanceolatum* to one of fine sands with *Ophelia borealis, N. cirrosa* and *S. bombyx*. Bare shingle habitat also became dominated by *Pomatoceros*. Peripheral impacts were at least equal to primary dredge No impacts were evident at distances of 1km or greater from the site of extraction. Emu 2002 Area 254 Cross Sands. Majority of area was characterised by mobile species and low diversity, abundance and biomass with the exception of an area of *Sabellaria*. Sand scour and sediment instability were identified as naturally suppressing the faunal assemblage. Sites within the currently dredged area were found to have depressed diversity and abundance relative to far field sites. Biomass was depressed across the entire area, the majority of which was contributed by polychaetes at both dredge and reference stations. Relatively elevated biomass was only apparent at sites with high occurrence of *Sabellaria*. No clear difference between sediment variables from sites within and beyond potential areas of dredging impact was observed in the 2002 study. A study in an adjacent licence area, (Emu, 2000) Area 436/202 indicated areas subject to intensive historic dredging had higher sand fractions, possibly as a result of overspill and screening. Dredge tracks were not evident within the area from SSS records indicating that these are rapidly in-filled by mobile sands characterising the wider sea area. Secondary impact sites were typified by development of mobile sand fauna reflecting the predominance of sands at these sites, although biomass was raised largely as result of large polychaetes such as *O. limacina* and *Nephtys caeca.* Millner *et al*, 1977. Areas 221 and 229 off Southwold. Despite considerable seasonal variation and variation between years during the study, abundances of the commonest fauna were considered to be depressed in the dredged area though it was not possible to state that this was due to dredging activities at the time of the study further analysis required. No impacts were evident or anticipated from outwash of fines due to natural high turbidity found in the area.	

ANNEX B

Table of Benthos Recoverability from Dredging Activity

Habitat and Community			
Static	**Shallow water mobile sand**	**Shallow water stable gravel with transient sand**	**Shallow/Deep water stable gravel**
Restricted area High intensity Low frequency	No recovery after 15 years was reported for sand pits in tidal flats at Terschelling (Dutch Wadden Sea). The pits were slowly filled with mud rather than sand. In highly dynamic subtidal areas, such as tidal channels, dredge pits were infilled and re-colonised within 1-3 years. The fast recovery of the biomass was attributed to the immigration of adults in tidal channels. In low dynamic subtidal areas, such as tidal watersheds, faunal recovery was slower lasting 5-10 years. Biomass was only 40% of the background after 4 years. The slow recovery of biomass indicated that adult immigration was less important in tidal watersheds.		Recovery regulated by the rate at which the seabed returns to the same physical condition. Deep pits fill very slowly in coarse gravel habitats, with the bottom of the pit accumulating fine sediments including muds and sands. Subsidence of coarse material occurs in combination with infilling by finer sediments, resulting in pits of reduced depth, but still with a sediment composition, seabed bathymetry and faunal composition unlike the original (Newell *et al*, 2002). Stable gravels support "equilibrium" communities with complex species-interactions and populated by comparatively slow growing species that may require a longer period of time for the recovery of species composition and biomass. In the low dynamic environment of the Danish Wadden Sea, recovery of the macrobenthos was reported to take 15 years. Certain components of stable gravels in the eastern English Channel are long-lived, slow growing and are recruited episodically. Dog cockles, *Glycymeris glycymeris*, in the eastern English Channel, for example, may be up to 14 years old with recruitment by juveniles at approximately 5 year intervals. Dog cockle re-colonisation and restitution of biomass may take as long as 15 □20 years.
Restricted area High intensity Medium frequency	Benthic recovery, following 1 □6 months dredging in the North Sea (Torsminde and Terschelling respectively) occurred within 2 - 4 years. Recovery of long-lived species abundance recovered within 2 years. Population structure took 4 years to recover. In low dynamic Mediterranean environments recovery of populations of commercially exploited bivalves may take 3 □10 years. Recovery was dependent on the duration of the impact of sand extraction.		
Restricted area High intensity High frequency			As above with no recovery over short periods of time.

Habitat and Community			
Trailer	**Shallow water mobile sand**	**Shallow water stable gravel with transient sand**	**Shallow/Deep water stable gravel**
Restricted area Low intensity Low frequency	(de Groot, 1979). Dredging for sand in the Gulf of Mexico (Alabama). Bottom recovery took 6 months with full recovery occurring within 2 years	Klaverbank: Abundance recovered fully within 8 months of cessation of dredging. Majority of species returned within 1 year. Large bivalves, which comprised a major proportion of infaunal biomass had not, however, recovered after 2 years. Newell *et al*, 2002. Area dredged lightly 12 months prior to survey and not at all before then, within 408 - Coal Pit. Dredged zone showed no significant suppression of species or abundance, the whole area being characterised by opportunists. k-dominance curves showed marginally higher dominance at dredged stations within 500m of activity. Dredge trails rapidly in-filled by transient sands. Recolonisation and restoration of community structure occurred within 12 months of cessation of dredging. Biomass was not significantly different to control stations after 12 months.	
Restricted area Low intensity High frequency	(ICES, 1979). Could not discern trailer dredge tracks 6 months after dredging. In some places, tracks that were 20-30cm deep disappeared within a few hours. (de Groot, 1979). Oregon (Pacific) Dredging for 8,000m3 sand. Bottom fauna had recovered within 28 days following. Fauna within the adjacent area affected by deposition of fine material recovered within 14 days (Lopez-Jamar & Mejuto, 1988). Faunal diversity, abundance and biomass increased significantly following dredging due to the import of opportunists. Within 3-4 months diversity decreased due to the increased dominance of the bivalve, *Thyasira flexuosa*. Highest similarity between pre and post fauna achieved within 6 months		
Restricted area High intensity High frequency			Kenny et al 1998, 3 years post dredging on experimental site. Taxa number has returned to baseline/reference as has the biomass. Numbers of individuals still reduced with respect to reference areas although similar to original baseline density. Transient sand/disturbance species characteristic e.g. *Dendrodoa* and *Balanus*. Biological recovery may become almost complete, with similar species composition, similar biomass and numbers of individuals substantially the same. Structural composition may differ, with continued dominance by disturbance species should sediment instability persist.

Habitat and Community			
Trailer	**Shallow water mobile sand**	**Shallow water stable gravel with transient sand**	**Shallow/Deep water stable gravel**
Large area Low intensity High frequency		Boyd *et al*, 2003b Area 222 20 miles east of Felixstowe. 25 years of dredging surveyed 4 years post cessation low intensity area. Sediments were similar to reference areas though were most similar to patches within an area of high dredging intensity where gravels were predominant sediment type. Overall, reference station sediments were more heterogeneous than low intensity sites with fines content equivalent to percentage content of sand in low intensity area. Sands were also present within dredged area as ripple formations possibly resulting from screened material deposition originating from both within and adjacent to area 222. Abundance, species number, richness, diversity and evenness all depressed in comparison with reference areas, though higher than for a high intensity dredge area within licence area 222. Biomass broadly comparable between low intensity area and reference stations.	No recovery while dredging continues, even with relatively low intensity dredging activity. Epifauna absent, continuing reduced diversity, abundance and biomass
Large area High intensity Low frequency			At the Hastings Shingle Bank recovery has been almost complete after 5 years without dredging, with balanced infauna and epifauna including key stability indicative species. Outer Owers, individual dredged tracts were deep, resulting in considerable delay in the recovery of the site. Tracks are still evident 8 + years later. Seabed area substantially returned to original composition, however, base of old dredge tracks populated by a reduced species assemblage typical of coarse sands including *Lanice conchilega.* Occurrence of *Crepidula fornicata* which is locally common on undisturbed areas, has led to some re-establishment of stability.
Large area High intensity Medium frequency			At the Hastings Shingle Bank reduced frequency dredging has resulted in improved diversity including the presence of epifauna with infauna, although infauna still most frequently present.
Large area High intensity High frequency	(Sardá *et al*, 2000). 150ha coarse-medium sand suction dredged between summer November 1994 in Catalonia. Rapid recolonisaton of the finer sediments by pioneer species within a few months. At the dredged site there was a 39% increase in total macro-infaunal abundance and a 37% increase in total macro-infaunal biomass following dredging over pre-dredge values. Diversity increased rapidly. Biomass showed rapid increase, attributed to the growth of echinoderm species, import and settlement of pioneers and elevated faunal densities immediately following dredging..	Newell *et al,* 2002. Area 408 Coal Pit. Site was heavily dredged until 36 months and lightly until 12 months prior to survey. Dredge trails rapidly in-filled by transient sands. Recolonisation and restoration of community structure occurred within 12 months of cessation of dredging. Biomass was not significantly different to control stations after 12 months. Non-dredged stations within the area potentially subject to deposition of screened material did exhibit suppression of biomass (by approx. 34%), though area beyond this was subject to elevated biomass (10 fold increase) in comparison to sites close to abandoned areas. This was indicative of organic material released during dredging or from benthic	No recovery occurring in high frequency dredged area still subject to dredging, with considerably reduced diversity and biomass. Little or no epifauna.

	Post dredged sediments were finer with restitution of the original coarser granulometry taking almost 1 year. Faunal densities returned to previous situation in two years although biomass was still elevated over pre-dredged values after 2 years. Diversity increased rapidly with most populations reaching a large size after 2 years although some species were still reduced. Populations of slow growing bivalves were predicted to take several more years to return to previous conditions.	plume increasing productivity in this area. (Desprez, 2000). 14 years of dredging (1980-1994) □16 months following cessation of dredging, species richness 100% restored. Abundance recovered to 56% of reference values and had stabilised by 28 months post dredging. Biomass showed slower recovery, 35% of reference values after 16 months and 75% of reference after 28 months. Community type had changed reflecting change in sediment type. Patches of mobile coarse sands colonised by *E. pusillus, Polycirrus medusa, N. latericeus, Syllis sp*. but also *Pomatoceros triqueter*. Where area was dominated by shingle and fine sand, community was dominated by sessile opportunists such as *P. triqueter* and hydrozoans and mobile opportunist species such as *Pisidia longicornis* and *Galathea intermedia* together with sand-dwelling fauna including *U. elegans, Cheirocratus sundervalli, S. bombyx* and *N. cirrosa*. The areas subject to peripheral impacts from screening/overspill in the depositional area were dominated by clean fine sands and *S. bombyx, O. bicornis, Tellina pygmaea* and *U. brevicornis* were the characterising species. Recovery in area subject to peripheral depositional impacts was slower with 60% decrease in species after 16 months, where the dredged area had recovered fully with respect to species number. Mean abundance was 86 % lower than reference and mean biomass values had only achieved 3% of the reference value after 16 months. Community was dominated by fine-sand species including *Tellina pygmaea* and *N. cirro*sa. There was a complete absence of gravel fauna and even species indicative of coarse sands were rarely recorded in this zone. Boyd *et al,* 2003b Area 222 20 miles east of Felixstowe. 25 years of dredging surveyed 4 years post cessation. □high intensity area. Dredged area contained higher sand and lower fines content than reference stations. Sands were also present within dredged area as ripple formations possibly resulting from screened material deposition originating from both within and adjacent to area 222. Abundances, species number, richness, diversity and evenness were all significantly depressed within dredge area compared with reference sites 4 years post dredging. Biomass significantly depressed within high dredge intensity area in comparison with reference sites. Community reflected higher sands contents at dredge stations. Juveniles were also recorded at higher abundance within dredged stations. Millner *et al*, 1977. Areas 221 and 229 off Southwold. Trailer tracks on seabed (0.3-0.5m deep) still visible 4 years after dredging had ceased.	

ANNEX C

Introduction to Remediation of Habitats

C.1 Example of Defining Conceptual Boundaries for Management Decisions (reference section 4.3.1)

Section 4.3.1 identified the need to define conceptual boundaries around the natural and anthropogenic elements to be considered in a given ecosystem remediation scheme.

The Pacific Northwest Coastal Ecosystems Regional Study (PNCERS) initiative has sought to bring managers and scientists together in a resource focused analysis of ecosystems. Central to this approach is the belief that communication between scientists (both natural and social) and resource managers is essential to the advancement of our understanding of the function and variability of coastal ecosystems, the effects of management practices, and the economic, social and ecological consequences of various types of change (PNCERS 1999).

At interdisciplinary workshops, the views of stakeholders have been used to assess the relative importance of "adverse" ecological effects and the relative significance of human activities and their contribution to these ecological effects. This assessment was focused on valued ecosystem components (VECs) specifically, Pacific salmon, Dungeness crab, Pacific oyster, and shorelands.

VEC diagrams such as that shown in figure C.1, are used to define the ecosystem boundaries and identify the range of influences that affect and / or depend on the ecosystem components. Remediation strategies which are rooted in a similar holistic conceptual approach will guide managers into considering, and hopefully addressing, the wider social and economic impacts of a given plan.

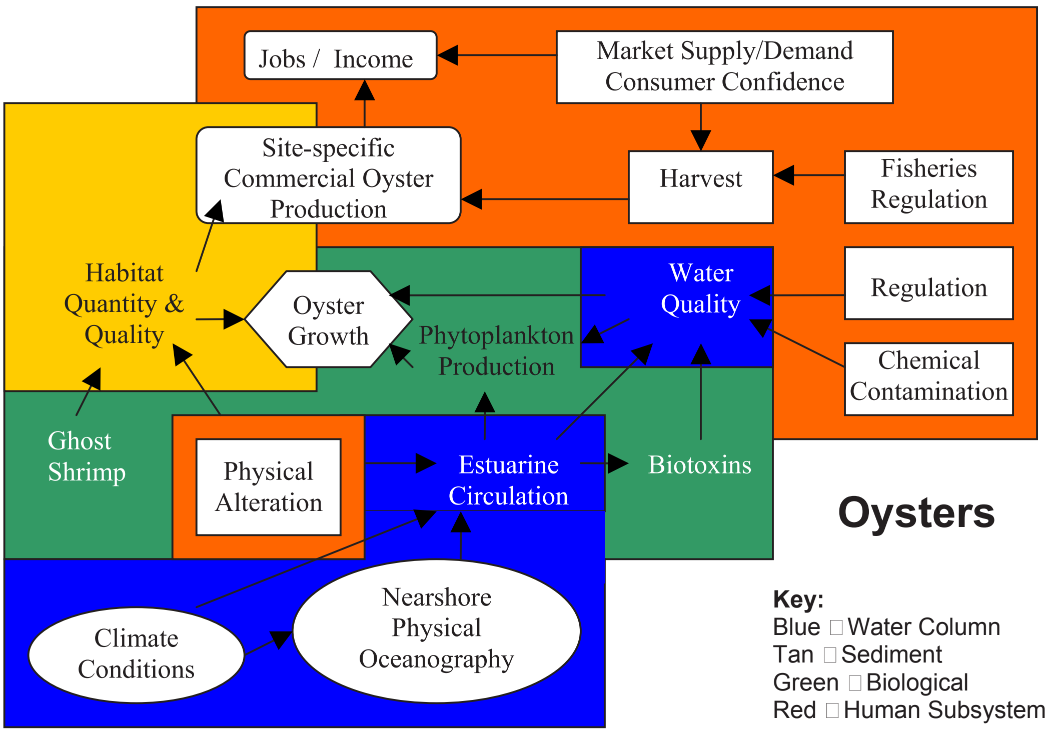


Figure C.1: Valued Ecosystem Components (VEC) for Pacific Oyster.
Source: Protecting and Restoring Pacific Northwest Estuaries (PNCERS 1999)

C.2 Policy Guidance on Restoration (reference section 4.3.2)

Statutory obligations are placed upon licence holders for terrestrial mineral workings concerning both the after-use and related aftercare conditions for extraction sites. These are contained in Schedule 5 of the Town and Country Planning Act (1990) and the Town and Country Planning (Scotland) Act 1997.

Aftercare Condition: A condition requiring that such steps shall be taken as may be necessary to bring the land to the required standard for whichever of the following uses is specified in the condition, namely agriculture, forestry or amenity.

An aftercare condition may either specify the steps to be taken, or require that steps be taken in accordance with a scheme (referred to as an aftercare scheme).

The steps that may be specified in an aftercare condition or aftercare scheme are planting, cultivating, fertilising, watering, draining or otherwise treating of the land.

Further policy on restoration and aftercare schemes for the terrestrial environment is contained within MPG7, the reclamation of mineral workings and in PPG9 (1994) Nature Conservation.

Paragraph 42 of PPG9 states that:

'*Extraction of minerals can create new types of habitat in areas where they were formerly rare or absent, while quarry faces may provide a valuable supplement to natural rock outcrops since features of geological importance may be revealed during quarrying operations. MPAs and mineral companies should bare in mind opportunities for habitat creation and enhancement even where nature conservation is not the primary end-use of the site*.'

Government policy thus provides considerable scope for habitat creation schemes for former quarries. Such government policy as exists regarding remediation / aftercare for the marine aggregate industry is contained in MMG1 which states:

'*Dredging should aim to leave the seabed in a similar physical condition to that present before dredging started in order to enhance the possibility of, and rate at which, the seabed recovers physically and biologically to its pre-dredging condition*.'

Compared to the situation on land, this offers less scope for habitat creation and it is not altogether clear what methods of remediation would be embraced by the above statement. This is determined on a case by case basis as part of the **conditions** which will be attached to new license applications. For approved licenses, the Secretary of State is likely to attach a clause specifying the condition in which the seabed is to be left at cessation of dredging. Any remediation undertaken for a given license area would need to operate within the parameters for aftercare laid down by the Secretary of State.

It is also possible remediation would be specifically ordered by the Secretary of State where monitoring of the dredge site or adjacent areas indicate an unacceptable impact.

'*Where monitoring indicates that the marine environment outside the dredging area is being unacceptably affected as a direct result of the dredging activity, the Secretary of State will consider carefully what action is needed to minimise further damage or, if considered necessary, to restore the area.*' (ODPM 2002)

C.3 Common Themes Marine vs. Terrestrial Explanation of Main Themes

Important parallels can be drawn between the quarrying industry and the marine aggregate industry which merit consideration in the development of remediation policy for marine aggregate sites.

C3.1 Physical Similarities between the quarried landscape / marine aggregate sites

- **Removal of overburden**
  Quarrying, in common with dredging, generally entails the removal of overburden. The removal of topsoil in quarries will generally result in a loss of fertility, which inevitably has an effect on ecological recolonisation. This characteristic should not be viewed as entirely negative since it tends to restrict invasion by aggressive nutrient demanding species, potentially allowing a wider diversity of less aggressive species to survive. However, where the objective for remediation is to recreate, as near as possible the pre-quarried landscape, the absence of topsoil will greatly extend if not completely impede this process.

- **The Creation of atypical landforms**
  Quarrying results in the creation of atypical landforms, with the disturbed land often being very different from the surrounding landscape. In the terrestrial environment, it is often these excavated landforms which are the very source of interest from an ecological or geological perspective, offering rare or sometimes unique topographical landscapes. Fifty nine percent of the Earth Interest SSSIs in the UK are currently former quarries (Wardell Armstrong 1996). Former quarries can comprise a whole range of features including mounds, hollows, pools, ledges and ridges which can provide important habitat niches (Wardell Armstrong 1996). The unique habitats which evolve at these sites are often the very reason for their SSSI designation. The corollary of this is that such sites are unsuitable for agricultural or other forms of development. Dredging, also results in the creation of atypical sea-bed forms, particularly in the case of static dredging depressions and to a lesser extent with trailer suction dredging. These can constitute unique interest as in the case of the Owers Bank pits (Section D3) but current UK licence stipulations require at least 0.5m of aggregate to be left on the seabed and therefore, there have been no cases encountered in the literature where underlying, hard substrate has been exposed. The other important parallel is that dredging pits can affect subsequent use of the seabed specifically fishermen whose trawling gear can be damaged by the uneven topography.

- **Instability of the worked environment**
  The excavations resulting from quarrying will invariably result in more unstable landforms. This applies both to exposed faces which can weather and erode and to spoil mounds which are prone to landslip. The instability resulting from marine aggregate dredging, while not a safety consideration as on land, does affect both recolonisation and the durability of dredging scars on the seabed. The emphasis with dredging has generally been to apply techniques which encourage the natural infilling of dredging tracks / anchor pits on the basis that restoration of the seabed topography will hasten a return to the surrounding ecological environment

Similarities in the Principles of Ecological Succession

Newell in his review of dredging impacts in coastal waters concludes that the general principles governing community structure following environmental disturbance appear to be generally applicable to a wide variety of communities both on land and on the sea bed (Newell et al. 1998). In section 3, the ecological succession from opportunistic species, which can withstand high degrees of stress, to equilibrium species, selected for maximum competitive ability in an environment that tends to be marked by biodiversity has been discussed. Similar principles apply to plant colonisation in the terrestrial environment. Grime identifies three primary plant categories:

- Competitors which exploit conditions of low stress and low disturbance;
- Stress tolerators which are adapted to high stress and low disturbance;
- Ruderals which are adapted to low stress and high disturbance.

Stress is seen as all factors limiting plant growth e.g. deficiency or excess of nutrients, gases, moisture, temperature and light. Disturbance constitutes destructive forces such as defoliation through grazing or burial through ploughing (Wardell Armstrong 1996 after Grime 1978).

The decision as to when and where to intervene in the natural recolonisation of a terrestrial site is driven by the objectives for remediation which might range from restoration of prime agricultural land to habitat creation to meet specific local BAP targets. There is value to be gained from reviewing how the principles of ecological succession have been harnessed in remediating terrestrial sites and applying the concepts to marine aggregate sites.

ANNEX D

Detailed Examples of Key Terrestrial Themes

D.1 Non Intervention / Natural recolonisation

There exist a large number of historic pits and quarries where excavation ceased many years ago, in which the natural processes of colonisation were allowed to proceed without interference and many of these now support habitats analogous to those on virgin land. Initial colonisation would probably have occurred from adjacent undeveloped land on the same substrate.

A fine example is Asham quarry, adjacent to Whatley quarry. While the land was purchased by ARC in the 1980s, the company has since surrendered its right to work the woodland area. Asham Quarry, itself has been colonised naturally from Asham Wood which represents one of the finest examples of ancient semi-natural woodlands in the Mendips (EN 1988). It is important to stress that while Asham quarry recolonised naturally, it is now subject to a high level of management with employment of a forester who undertakes coppice management, tree felling, maintains fencing against deer etc.

Non-intervention is rarely an option if nature conservation objectives are to be met. This is because the residue of undeveloped natural land is significantly smaller than it was before World War II. Accelerated agricultural and infrastructure development in the post war years has significantly altered the colonisation of quarries where work has ceased. Habitats are often likely to be colonised by a different range of species than those once typical of the substrate. Biodiversity initiatives seek to preserve endangered habitats which were once a common feature of the landscape and this frequently involves restoration of ecosystems through active management.

Figure D.1: Natural Recolonisation of Quarry. Source: Kids Ark, Gloucestershire

D.2 Restoration of Ecosystems

Complete restoration of an ecosystem to a pre-excavation state in the terrestrial environment is not a feasible option. Since excavation removes huge quantities of substrate, which cannot be put back, inevitably this affects the fundamental characteristics of the ecosystem the nature of the underlying soil and rock, drainage, nutrient supply, sunlight etc. and the resulting flora and fauna. It is recognised that substrate plays a dominant role in habitat determination and exposed rock can offer opportunities to restore former habitats.

It is the very rarity of habitat found in some of the naturally recolonised quarries, which has made them targets for nature conservation and also vulnerable to invasive species which threaten their existence. Calcareous grassland habitats have developed at many former limestone quarries, hosting a rich variety of flora and fauna which includes orchids and other attractive flowering plants. Many have SSSI status.

It is often the early colonisation species which are particularly significant, including the mosses and lichens which develop on the bare limestone substrate. Left unmanaged, invasive species, typically willow, hawthorn and scrub birch will begin to threaten and dominate the herbaceous plant communities, resulting in their demise. Management of these communities focuses on preserving the rarer earlier colonising species. Examples of these sites include the Portland and Masters sites in Dorset, Inmothsay quarry and Wharram quarry.

D.3 Topographical Rehabilitation

Topographical rehabilitation in the context of this study is using material to fill an excavation once it has been worked. There are important lessons that have been learnt from early experience with this form of remediation which are of direct relevance to recent policies to use former static dredging or borrow pits for dumping waste (see section F.3(3)).

During the post-war period, most restoration has been to agricultural land not wildlife habitat (RSPB 2003). This has generally required infilling to level the landscape and it has proved expedient to use waste to achieve the landfill. Prior to the Review of Old Mineral Planning Permissions (ROMP) under the Environment Act 1995, the aftercare conditions attached to many older permissions was for sites to be used for landfill and subsequent landscaping / return to agricultural use.

Until the mid 1970s there was very little awareness of the long-term problems of indiscriminate dumping at landfill sites. As a result, dumping of waste at former quarries was largely uncontrolled and lacking in technology to minimise and contain the problems of leachate.

Minley Quarry in Hampshire was excavated for filling soils used in the construction of the M3 motorway in the early 1970's. The site was identified for possible landfilling with municipal solid and amenity wastes. The landfilling was approved in the era of COPA 1974 and Planning Resolutions implemented on the basis of the dilute and disperse principle which was prevalent at the time. The restoration probably comprised sub soils and topsoils to cover the wastes and to a profile to approximately match surrounding topography. The site appears not to have attracted any afteruse. The NRA as predecessor to the Environment Agency had observed the appearance of a brown "sewage fungus" in a nearby watercourse for many years, but the association of the proximity of the landfilling appears not to have been clearly established.

The Environment Agency later initiated relatively complex works about 20 years after restoration under powers of the Water Resources Act 1990 to restrict impact of the dilute and disperse landfilling. The life span of the works and its management is also unclear, but likely to be 5 -10 years duration.

On both construction and operational resource issues and the matters of uncertainty due to complex hydrology, the case illustrates the level of risk of long-term problems attendant with remediation when disposal options fail to adequately address the environmental impacts of a proposed scheme on the wider ecosystem.

D.4 Habitat Creation

It is evident from what has been written above that nearly all forms of remediation involve some form of habitat creation or enhancement, albeit that the objective may be to recreate former habitats which are now rare. There is a distinction between the approaches outlined above, in which the focus is habitat management of the baseline ecological conditions, and habitat creation schemes which specifically set out to develop a habitat for a species or community of species which did not previously exist at the site.

The use of quarries for habitat creation forms an integral part of government policy for nature conservation. Planning Policy Guidance Note 9 (PPG9) paragraph 42 states:

'Extraction of minerals can create new types of habitat in areas where they were formerly rare or absent, while quarry faces may provide a valuable supplement to natural rock outcrops since geological features may be revealed during quarrying operations.'

In the terrestrial environment, habitat creation has several important roles (White, GJ and Gilbert, J C 2003):

- Contributing to increasing biodiversity and the achievement of BAP targets;

- Linking fragmented areas of semi-natural vegetation in an otherwise intensively used landscape;
- Buffering existing semi-natural areas;
- Expanding existing sites, so making their management more practicable or economically viable;
- Providing new areas which can serve as local amenity and educational resources and which can draw people away from fragile habitats.

Habitat creation in the marine environment which is discussed further in section 5.4 would need to be guided by some of the above objectives, principally the need for schemes to be underpinned by biodiversity targets.

The range of options for habitat creation is as broad as the technical and cost constraints will allow and projects can vary in scale from relatively low cost projects such as the construction of bat caves at Stowfield quarry in Gloucestershire to large scale projects, for example the creation of a periglacial valley at Sandy, Bedfordshire or the 700 acre wetlands project associated with Needingworth quarry in Cambridgeshire.

Some examples are discussed in greater detail below, specifically in relation to principles which may apply in the marine environment.

Case Study - Stowfield Quarry

Bats are a target species within the UKBAP. Mineral extraction sites with their diverse micro topography can provide suitable habitats for bats which rely upon a mosaic of corridors which connect roosting sites and feeding sites in an area relatively undisturbed by humans. Natural fissures in quarry faces offer roosting sites. Where no natural roosting sites exist, these can be created using concrete pipes and grills as has been undertaken at Stowfield Quarry. This represents an example of low cost habitat creation requiring little management which has been specifically linked in with the local biodiversity plan and targets.

Figure D.2: Horseshoe Bat
Source: The Bat Conservation Trust

Case Study - Needingworth Quarry

The objectives of sustainable development and biodiversity are often seen to be in competition in many planning issues. Needingworth quarry and wetland project represents a success story for integrated planning between Hanson Aggregates and the RSPB, which will ultimately create 700 hectares of wetland as well as 30 million tonnes of aggregate over a thirty year period. Some key lessons can be drawn from this case:

- **Realisation of a nature conservation opportunity**
  The original planning permission submitted in 1993 presumed restoration to agricultural land post extraction. Subsequent surveys demonstrated the lower quality of the agricultural land and RSPB together with other conservation bodies proposed the creation of a wetland habitat because of the inherent suitability of the site, namely the low lying land in relation to the elevated level of the River Ouse and the opportunity to extend habitats established at existing adjacent conservation sites Berry Fen SSSI and Ouse Washes SSSI, Ramsar Site and SPA.

Figure D.3: Reedbeds
Source: Minerals Nature Conservation Forum.

- **Clear objectives linked to a BAP**
  The UK BAP target is for the creation of 1200 hectares of reedbed on land of low nature potential by 2010. Needingworth, by incorporating 360 hectares of reedbed will go a long way towards meeting this target.

- **Extensive consultation in the planning stages**
  It is only through appropriate consultation between the industry and the RSPB that such an ambitious scheme has come to fruition. Extensive consultation with the local community was also undertaken to involve them in the planning process and ensure that it met local amenity needs as well as nature conservation objectives.

- **Detailed long term planning**
  The complexity of design required detailed planning which included the need to forecast water usage for both wet and dry years and accommodate extraction activity with wetland development. The latter is being achieved by a modular approach with 20-40 ha. of wetland (one or two years worth of extraction) being created each year.

- **Provision for long-term monitoring and management**
  The RSPB will be involved in both monitoring progress with the project and the long-term management.

- **Exploiting other community benefits**
  There is the potential for ‘wealth creation’ through harvesting and sale of the reeds to help offset management costs for the site as well as the development of educational resources for raising awareness for biodiversity and support for conservation.

All the above merit consideration in any discussion of remediation schemes for the coastal or marine environments, albeit not all are applicable in every scenario.

ANNEX E

Detailed Examples of Key Coastal and Intertidal Themes

E.1 Coastal Shingle Extraction Habitat Restoration

The extraction of sand and gravel from a beach or shingle structure can be damaging and can lead to a loss of coastal vegetated shingle habitat, create depressions in the shingle structure, and in cases where excavation occurs to a level below the water table create flooded gravel pits. Several sites in the UK have been adversely affected by this activity including The Crumbles, East Sussex and Dungness, Kent (Doody and Randell 2003).

Attempts have been made to restore the degraded shingle habitats specifically in terms of restoring vegetation; this has included the restoration of the shingle matrix and the topography of shingle structures. In Orfordness, Suffolk one of four major shingle landforms in Britain, the shingle morphology had become severely degraded and considerable damage and loss of vegetation had occurred; this can be attributed to a number of activities including shingle extraction for beach recharge and military activity. At Orfordness in 2000 a test was undertaken by the National Trust to ascertain whether it was possible to regenerate shingle flora on some of the worst degraded and damaged sites. A test site was selected in an area significantly degraded by military use. Initial work on the ridge to be restored involved the scraping off of the surface shingle to a depth of approximately 20cm, and the shingle graded into four size ranges representative of the natural sizing on an adjacent shingle ridge. The material was then replaced onto the scraped area in order to reproduce the height, width, spread and size ratio of the nearby undamaged ridge; in some sections fines and seed were also added. Though morphology of this section of the ridge was restored the re-establishment of vegetation has not readily occurred even with introduced seed as no evidence of recolonisation was found 2 years after the completion of the works.

Aggregate extraction from these coastal structures may not always be destructive, shallow excavations which do not reach the water table can create a new wetter habitat and become recolonised with vegetation and may increase the biodiversity of the site. In addition, excavation to below the water table can create valuable wetland habitat and saline lagoons. For example excavated gravel pits in Dungeness are now owned by the RSPB and a large part of these have become a wetland nature reserve (Doody 2003). Several of the gravel pits here have also been regraded around their edges to give a shallower gradient more suitable to certain vegetation communities and some have become home to rare invertebrates. These artificial wetlands and saline lagoons have also helped to achieve Biodiversity Action Plan (BAP) targets, for example new lagoons at Orfordness have contributed to Suffolk's BAP which aims to increase the area of lagoons in the county.

Figure E.1: RSPB Reserve Dungeness, Kent
Source: Hanson

E.2 Restoration and Creation of Intertidal Mudflats and Saltmarsh

The restoration of coastal intertidal flats and salt marshes can be undertaken by three methods. The first method is **coastal realignment** which involves the landward relocation of a maintained flood defence and the restoration of intertidal habitats by the reactivation of the coastal flood plain, and has been undertaken as several sites in the UK particularly in Essex (ABP 1998). This method has no direct parallels or principles in relation to the restoration and enhancement of subtidal habitats and will not be considered further in this document.

However the other two methods, **enhancement sedimentation** and **foreshore recharge** do have some parallels with the restoration of subtidal habitats and are discussed below.

1) Enhanced Sedimentation/Polders

Where intertidal erosion is a problem and there is a high content of fine sediment in suspension, it is possible to encourage the settling out of sediment by increasing the time of slack water and reducing tidal and wave action. This method has been used in Holland and Germany to create saltmarsh on tidal flats and has been used extensively in the Wadden Sea.

Figure E.2: At high tide, water fills the dyke (from the left to the right of the picture) and is trapped by the earth barrier when the water level drops.

The simplest method is to create an earth dyke that fills with water at high tide and slowly drains out through porous sides on the ebb, leaving any sediment that was in suspension trapped in the bund (Figure E.2). This approach can be enhanced by digging a series of trenches and piling the excavated mud between the trenches, this is also called 'gripping' and helps to develop plant colonisation on the raised areas and trap sediment in the trenches.

This process can be enhanced further by the addition of brushwood groynes and wave baffles. Brushwood structures are permeable structures usually composed of bracken or old Christmas trees, which impede the passage of water sufficiently to allow sediment to accrete. Wave baffles are used to reduce tidal currents and wave action in an area, and may reduce currents sufficiently to allow sediment to settle. Wave baffles have been constructed from old Thames lighters in Essex and are often associated with groynes connecting the wavebaffle to the shore, thus creating a shelter quadrangle. These enhanced accretion techniques, even if successful, can take 30-40 years for sufficient sedimentation for saltmarsh to become fully established.

This method has been applied to several sites in Essex including a site off the south east coast of Mersea Island in the outer reaches of the Colne Estuary, here five brushwood enclosures have been built on the foreshore, and some evidence of accretion has been found in sheltered parts of the construction (Doody 2003).

A similar experiment for saltmarsh restoration has also occurred in three locations along the Dengie peninsula, Essex:

- Sales Point, 11 Thames lighters were placed 200m offshore spaced 20m apart to create a wavebreak in 1986 protecting 600m of eroding saltmarsh, and in 1989 shore to lighter connecting brushwood groynes were erected.

- Marsh House, 16 Thames lighters were placed 500m offshore, spaced 20m apart in 1984. Two groynes were also constructed clad with geotextile material rather than brushwood at either end of the wavebreak, *Spartina* was also planted in areas to the landward side of the Polder.

- Deal Hall, two 400m^2 polders were constructed in 1980 using double row wooden stakes infilled with brushwood fence. On the inner side of the groynes sediments excavated from a shallow ditch parallel was heaped to stiffen the structure and make the lower level impermeable to water; the sites were gripped in 1981 and 1989.

At Deal Hall significant accretion within the sedimentation fields was reported, and at Marsh House the elevation of the mudflat within the sedimentation field has been raised and is maintaining a constant equilibrium elevation above that of the adjacent natural mudflat and below the level for saltmarsh development (Atkinson et al 2001). However recent inspection has suggested that these measures have yet to make any difference to the establishment of new saltmarsh (Doody 2003). Elsewhere in Essex the success of such sediment fields has been mixed with several failures, the techniques are believed only to be successful where the local sedimentary trend is towards accretion.

2) Foreshore Recharge and the Beneficial use of Dredge Material

There are two basic methods utilised for recharge of intertidal flats and saltmarsh; **trickle charging** where the sediment is placed at a single point or at a series of points on the lower foreshore or just offshore from the target intertidal area, allowing wave action and tidal currents to redistribute the sediment; and the pumping or spraying of sediment directly on to the target area usually from a dredger moored close to shore. The spraying of sediment onto the intertidal area is often termed **rainbow charging**.

Figure E.4: Dredger Rainbow Charging
Source: BMAPA

Examples of beneficial uses of dredging for habitat restoration and creation and the methods used are given in table C.1 below. The direct placement of sediment on to an intertidal area by heavy plant such as the methods used for beach recharge is not suitable, due to the likely damage to saltmarsh and the soft nature of mudflats.

Location	Date	Habitat Restoration or Creation	Comments
Horsey Island, Essex, UK (ABP 1998)	1990	Intertidal recharge	The objective was to recharge an intertidal area with coarse dredging material to protect Walton Backwater from erosion and flooding. It was decided to use coarser sediment rather than the sediment naturally present, as it would be more stable and less susceptible to erosion, reducing the risk of the material moving offshore and being lost from the recharge site. 1800m^3 of dredgings were taken from Harwich Harbour and sprayed on to the mid intertidal outside an abandoned seawall, the recharge material was a mixture of coarse sands, pebbles and grits, the recharge site consisted of fairly uniform muds. Monitoring revealed a complete charge in sediment character due to the recharge using coarser material; benthic surveys revealed a marked change in the marine invertebrate communities. Abundance of mud dwelling communities was initially reduced by smothering; the altered composition resulted in colonisation by species not found elsewhere on the site. The recharge site was rapidly colonised by benthic invertebrates particularly *Nereis virens*, which helps support the local bass fishery and bird populations. A new marsh habitat is now developing behind the recharge.
Pewit Island, Blackwater Estuary, Essex, UK (ABP 1998)	1992 & 1995	Intertidal recharge	Coarse dredging material was placed in a mound on the intertidal foreshore in two phases 2529m^3 in 1992 and 2646m^3 in 1995. Non-cohesive sediment was used with a wide grading curve sourced from the maintenance dredging of the Harwich Haven and a rainbow discharge was used to deposit the material. It was found that the material moved up the foreshore and inhibited the erosion of the saltmarsh cliff behind. However, the coarse materials supported a reduced invertebrate biomass and therefore a decrease in potential food supply for waders. Some of the existing parts of the saltmarsh behind became smothered by the roll back of the sand and gravel ridge over the edge of the salt marsh. Fine material is beginning to be deposited over the top of the coarser material in the lower intertidal.
North Shotley and Trimley, River Orwell, Suffolk, UK (Doody 2003)	1998 & 2001	Intertidal and saltmarsh recharge	High density slurry was used to facilitate the creation of a marsh surface with variable topography. At Trimley, channels were excavated onto the existing surface prior to recharge with high density slurry with the aim of establishing whether pre-engineering enhances creek development. The dredge material had both a positive and negative effect on the invertebrate fauna; the deposited material initially smothered the existing benthic communities, but enhanced the invertebrate fauna thereafter.
Calcasieu Ship Channel, Louisiana, USA (Peyre *et al* 2002)	1999	Saltmarsh restoration	Terraces have been constructed and have reversed the shoreline erosion and created 17 acres of saltmarsh with an interface of 1500m. Terracing is a method for restoring saltmarsh edge habitat and is used to replace marsh and encourage sediment accretion in surrounding open water areas. Terraces are built in open water areas of the estuary where marsh previously existed by piling dredgings to form a discontinuous linear ridge which floods at high tide, these ridges are also planted with fast growing marsh plants e.g. *Spartina alterniflora*. Multiple terraces are built in a pattern to maximise marsh edge and decrease wave energy.
Maldon, Blackwater Estuary,	1993	Saltmarsh restoration	The area was experiencing a decrease in active saltmarsh and breaching of the salting separation between Heybridge creek and the main channel, it was feared that this pattern of erosion would result in

Location	Date	Habitat Restoration or Creation	Comments
Essex, UK (ABP 1998)			loss of depth at Maldon Quayside and accelerate the loss of the remaining marsh. Cliffing of the saltmarsh edge had also rendered the saltmarsh banks susceptible to undercutting and mass failure. The gaps in the salting between the creek and river channel were plugged with wooden planking and timber piles and infilled with cohesive dredgings. Dredged spoil was deposited in front of the eroding edge of the saltmarsh in order to create a sloping rather than cliffed bank profile, the edge of the saltmarsh was also extended through the deposition of dredged spoil. The closure of the breaches was successful and tidal flow through them eliminated, there is also evidence of re-colonisation of the areas with saltmarsh vegetation. The cliff stabilisations appear to be successful, however on the extended saltmarsh little re-vegetation has occurred; it is believed the elevation of the new mudflat is too low.
Horsey Island, Essex, UK (ABP 1998)	From 1990	Saltmarsh recharge	A site located off south-east corner of Horsey Island was suffering from vertical erosion of saltmarsh. Half a load of dredged silt was sprayed on a 0.5ha plot, of heavily grazed saltmarsh, which was above mean high water. The work was carried out in August so that consolidation of the sediments could take place before the main release period for the saltmarsh plants seeds and the arrival of overwintering bird species. It was originally thought that most of the silt applied would wash off the site over the first spring tide. There was no apparent loss of plant cover, some sediment remained in the depressions on the saltmarsh surface and was quickly covered by vegetation. More sediment may have been retained if the site had been bunded; this may also have enhanced accretion further.
Horsey Island, Essex, UK (DECODE 2001)	1998	Saltmarsh recharge	The environment agency recharged 20,000m^3 of cohesive sediment from Harwich Haven between a shingle berm and sea wall. After nine months considerable saltmarsh growth specifically *Salicornia sp.* had occurred over parts of the recharge area. The area was again recharged during January 2001; the aim being to raise the height of the mud surface to allow the establishment of higher saltmarsh plants.
Medway Estuary, Kent, UK (ABP 1998)	1996	Saltmarsh recharge	The site is located on the intertidal flats around Bedlam Bottom in the Funton Creek, a total 4000m^3 of fine dredge materials were placed on the lower intertidal and left for natural hydraulic processes to gradually move it up the foreshore (trickle charge). This approach enables sediments to be redistributed within the intertidal system and promote natural evolution of the intertidal habitats. Early results indicated that bottom dumping sediments and trickle feeding is a success for relatively small infrequent volumes and approximately 50% of the material was estimated to have been retained at the recharge site.
Suffolk Harbour, Orwell Estuary, Suffolk, UK (DECODE 2001)	-	Saltmarsh creation	Changes in the river have resulted in a loss of intertidal mud level in the Orwell adjacent to the harbour. The dredged material is very fluid and pumped on to the foreshore and placed within wattle hurdles or faggots. Sufficient material has remained at the site to raise the tidal height of the foreshore to allow saltmarsh plants to colonise.

Location	Date	Habitat Restoration or Creation	Comments
Titchmarsh Marina, Walton-on-the-Naze, Essex, UK (DECODE 2001)	1998-2001	Saltmarsh creation	The marina suffers from regular deposition of fine sediment and needs frequent dredging. To allow for intertidal recharge of fluid dredge material, excavated earth was used to construct a bund on the west side of the marina, and sediments from the marina were dredged and pumped into this recharge area. During 1998 and 1999 10,000 tonnes of mud were pumped into the bunded area. More mud was added in 2001 to raise the marsh height further and allow saltmarsh plants to develop.
Black Rock Harbour, Connecticut, USA (ABP 1998)	1983	Saltmarsh creation	A 7000m^2 area of saltmarsh and wetland was created by dumping dredge material over the area and excavating to the desired height, with a water depth of 0.3m at mean high water. The site was then planted with *Spartina alterniflora*. Plant growth was poor in the first year, but vigorous four years post-construction and there is now a dense vegetation stand and subsequently a diverse benthic community and associated bird population.
Sheep Island, Maine, USA (ABP 1998)	1988	Mudflat creation	Silty sands from maintenance dredgings where placed on a sheltered 1.2ha site of shallow sub-tidal sand and gravel to create and intertidal flat. Monitoring occurred in 1990-92 and it was found that a diverse infaunal community had developed on the new intertidal mudflat.
Parkstone Yacht Club, Poole Harbour (ABP 1998)	1995	Mudflat creation	As part of the planning consent there was a requirement to provide an area of intertidal mudflat to replace that lost by development. The mudflat created was 325m long by 20m wide. The mudflat was built on the inside of a rubble mound breakwater which protects the marina and sheet piling which is at a depth of +1.2CD approximately mean low water neaps; the edge of the break water is at a depth of +2.0m CD approximately mean high water springs. Dredgings from the marina were used to construct the intertidal mudflat and a total of 7000m^3 were used for the initial fill and 3000m^3 from the existing intertidal prior to construction were used for the top layer. Surveys undertaken in 1996 showed good initial flora and fauna populations.

Table E.1: Examples of the beneficial use of Dredge Material for the Restoration and Creation of Intertidal Habitats

ANNEX F

Detailed Examples of Key Marine Themes

F.1 Marine Protection Areas Active Passive Management

The concept of Marine Protection Areas (MPAs) to conserve fisheries and the marine environment has come largely from sedimentary fish living on trophic reefs (Polunin and Wabnitz 2001). In Australia, New Zealand and the Seychelles, MPAs have been in use since 1970s, where they are a key part in Government strategy for conservation, marine environmental management and promoting ecotourism (Jennings 1999). France, Spain and Italy have set up MPAs in areas where natural resources have diminished, with the aim to restore and revitalise downgraded marine habitat. These MPAs are also aimed at developing ecotourism specifically scuba diving, allowing marine protection to go hand in hand with a viable tourist industry (Kyriakopoulos *et al* 2002).

A significant amount of the research into MPAs has focused on the conservation and protection of fish stocks, however restricting these activities will also have an effect on the seabed and benthos. Certain types of fishing gear can have a significant impact on the seabed; trawl doors scrape the seafloor and can penetrate the sediment up to 15cm, while beam trawls can penetrate to a depth of 8cm across the entire width of the beam. Organisms directly in the part of the gear maybe caught as by-catch, while burrowing species may be crushed (Polunin and Wabnitz 2001, Jennings 1999).

A review undertaken by Polunin and Wabnitz (2001) concluded that sheltered locations with naturally low levels of disturbance and sensitive habitats will benefit from protection from trawling. But in shallow waters and exposed locations where natural disturbance by tides and wave action is high, habitats in trawled areas are not expected to differ significantly from those protected from trawling. Appropriately designed MPAs can act as a control of exploitation effects, but only with respect to relatively sedentary organisms and habitats adversely affected by fishing. Therefore, in stable deep water areas MPAs may help to enhance recovery rates of benthos after dredging, but in more changeable areas such as mobile sands banks MPAs may have little or no positive impact.

F2 Examples of Preserved Seabed Habitats

Habitat mapping was completed for an area off Shoreham, using BGS geophysical data and biotope data collated from grab samples, trawls and video evidence. Video and still camera surveys of the former dredging pits on the Inner Owers Bank showed the sediments and benthic assemblages within the former dredging area to be very different from adjacent areas. In particular, the bases of the pits were found to harbour dense communities of *Mytilus edulis* (common mussel) together with the branching bryozoa *Pentapora foliacea*.

Figure F.1: *Pentapora foliacea* (Ross Coral) Source: Seasearch website.

Experience from the quarrying industry has demonstrated the unique value of former mineral extraction sites, either on grounds of the unique geology or habitats created. As policy is developed to consider what remediation, if any, should be undertaken at marine aggregate sites, further investigation should be made into the potential intrinsic value of some former dredging sites.

F3 Examples of Marine Restoration Projects

1) Roughs Tower Disposal Site Harwich Haven Authority

Harwich Haven Authority provides pilotage, dredging and navigation services to the five Haven ports of Felixstowe, Ipswich, Mistley, Harwich International and Harwich Dock Company. The Harwich Harbour Act 1974 gave the Authority the power to dredge the seabed to maintain and improve navigation.

The progressive development of Felixstowe in conjunction with the increasing draft of container vessels has resulted in a succession of channel deepenings, the most recent occurring between 1998 and 2000. These were subject to the Conservation (Natural Habitats etc) Regulations 1994.

Attached to the 1998 consent of the capital dredging application was a mitigation package which included remediation for the Rough Towers disposal site. This involved placing a stiff clay bund around the disposal site to contain the material and then capping the surface with gravel to create a stable habitat for crustacea. To prevent damage to existing and newly created crustacea habitats, maintenance dredgings are taken to a new dispersive disposal ground further to seaward. Dispersion of fine silts avoids impact on the seabed and distributes the material using natural forces (DFT 2003).

The original plan for a 50mm layer of gravel was subsequently modified to a 25mm sprinkling.

A separate application was made by the Kent and Essex Sea Fisheries Committee for funding to support a Lobster Stock Enhancement project under the DEFRA Financial Instrument for Fisheries Guidance (FIFG) grant scheme (Kent & Essex Fisheries 2002). The project has two elements: firstly, a V-notching programme to identify, mark and so protect mature female lobsters to assist with the generation of a brood stock; secondly, the rearing of juvenile lobsters taken from local parent stock at the National Lobster Hatchery at Padstow for subsequent release back onto local grounds. Provision of the female lobsters has been possible through a buy back scheme agreed with local fishermen.

The four year scheme which commenced 2003 has experienced problems from the outset. The initial batch of berried lobsters supplied to the hatchery died soon after arrival, predominantly attributed to plant inadequacy during the hot summer. Intervention in this instance has proved counter-productive to date, although the intention is to learn from the initial lessons and continue the project.

There is a further potential concern that lobster larvae spend a long time in the planktonic stages. As they grow, they tend to exhaust the food supply and progress to increasingly larger crevices within a natural or artificial reef structure. Because the distribution of crevices in reefs is fractal in character, i.e. there are far more small crevices than large ones, a lobster can experience a decline in the number of suitable sized shelters as it grows and is therefore compelled to migrate to another locality. Strong currents and predators can result in high mortality during migration. (Jensen Wickins & Bannister 2000). Use of DNA fingerprinting to gain further insight into the migration of juveniles is currently the subject of research being undertaken at Belfast University (J Wiggins personal communication).

Results of monitoring undertaken by CEFAS staff are still awaited but anecdotal evidence suggests the gravel layer on the Roughs site has been successful in attracting back lobsters. Such findings are consistent with the general finding that lobsters can occupy an artificial structure within a very short period of time following deployment (Jensen Wickins & Bannister 2000). Some by evidence of colour rings have travelled significantly further distances than had been anticipated.

This case illustrates a number of important points in relation to marine site restoration:

- Use of the Conservation (Natural Habitats &c) Regulations in relation to offshore development;
- The importance of local consultation in influencing the mitigation package for the capital dredge and disposal arrangements.
- The application of gravel seeding with initial encouraging results.
- The potential adverse effects which can result from intervention in community structure.

2) Remediation of Drill Cutting Piles

The identification and selection of remediation options for drill cutting piles is a key issue faced by the offshore oil industry in relation to platform decommissioning. Current evidence suggests hydrocarbons are the principle cause of environmental impact from drill cutting piles although other factors which could cause some of the observed effects from drill cuttings piles, such as the physical impact of smothering, changes in sediment grain size, organic enrichment and other possibly toxic components of drilling muds (CER 1999).

This issue is considered to be of interest to this study for the following reasons:

- Discounting the hydrocarbon element, drill cutting piles potentially pose secondary physical problems to the benthos which are similar to those resulting from deposition of dredging / screening spoils;
- Drill cutting piles along with marine aggregate extraction sites are subject to the same 'in combination' impacts, principally from the trawling industry;
- Some of the remediation technology has potential application for marine aggregate site remediation;
- There is common ground in terms of the stakeholder issues, and consequently the consultation approach could offer a basis for consideration by the marine aggregate industry.

Figure F.2: Oil Rig
Source: Outer Continental Shelf (OCS) Web site

Physical impacts of drill cutting piles

Drill cutting piles constitute concentrations of fine sediment ejected from the shale substrate during the drilling process but are also mixed with the drilling muds used to lubricate the drilling process (which can contain up to 80% oil).

In terms of impact assessment, it is not possible to isolate the physical impacts of smothering from the toxic effects of the hydrocarbons. In common with the natural sinks which can form downstream of dredging extraction sites (Possford Haskonin 2002) drill cutting piles are subject to the natural processes of either weathering and biodegradation or waves and currents, particularly those related to storm activity, which may suspend the sediments from the centre of the cuttings pile and distribute them onto the surrounding seabed.

If sediment redistribution is dominant, the affected area will expand over time. Which process dominates will depend upon the physical properties of the sediments in the pile and the strength of bottom currents in the area and will therefore be site specific (CER 1999).

The physical characteristics and stability of dredging spoil piles will be subject to similar processes and therefore the techniques used to stabilise drill cutting piles are considered here.

In combination impacts

Both drill cutting piles and marine aggregate sites are subject to ‘in combination’ effects in relation to the trawling industry albeit the nature of these effects is very different. For drill cutting sites, the rate at which contaminated material disperses outwards may be increased if piles are disturbed by fishing activity after the platforms are removed. This could potentially significantly extend the area of impact.

For marine aggregate sites, based on studies of the effects of trawling on community composition, it is reasonable to suggest that trawling of former aggregate sites could adversely affect the rate of natural recovery of the benthic community (see F.1 paragraph 2). Government policy contained in MMG1 acknowledges the impact of aggregate dredging at one site has the potential to interact with impacts from other sites in close proximity, or with other human activities such as fishing, pipeline discharges or the disposal of harbour dredgings. These require consideration within the EIA process (ODPM 2002)

Furthermore, both drill cutting piles and dredging scars, notably anchor pits, represent potential snagging hazards for trawler gear and as a result are a source of concern to the fishing industry.

Remediation Technology Potential Applications to the Marine Aggregate Industry.

Foremost in remediation techniques for drill cutting piles is the aim of preventing the high concentration of hydrocarbons from contaminating the local seabed ecology. Within the inventory of techniques are removal, entombment and capping of the pile. Of these, only capping has a potential application to aggregate site remediation.

Capping

Capping of drill cuttings can involve the placing of concrete mats with an impermeable synthetic membrane over the drill cuttings. The mats measuring 5m x 4m would be designed to be joined together by divers and be anchored where they contact the seabed. (Cripps *et al* 1998).

Capping was not favoured by the Independent Review Group for decommissioning of the North West Hutton platform. Capping was seen as postponing rather than solving the problem since no capping will be permanent. Logistically it is complex and expensive about 600 mats would be required for the 120m diameter North West Hutton pile and there are also issues related to residual liability and impacts on other users of the sea that have to be considered for in situ options (CER 1999).

It has also been suggested that gravel dumping could be undertaken around the pile to ensure the membrane remained in situ. Frond mats could be installed around the perimeter of the pile to encourage the development of natural ecology. The method is likely to be fairly expensive and will probably only be used in sensitive zones (fish spawning areas) or if the pile is unstable (leaching). If the intention of the covering is to protect the pile from physical damage, other options will probably be more suitable.

Capping with concrete mats will create an artificial environment very different from the surrounding sea floor. This will be colonised by organisms typical of hard substrata, producing a community greatly different to the usual sediment dwelling benthos (CER 1999).

The above considerations are pertinent to the debate on the appropriateness of depositing any form of artificial structure on the seabed (see also paragraph F4.2).

Gravel dumping

Gravel dumping is an established technique in the offshore industry, used for many applications including adding a protective cover to exposed or free-spanning pipelines or structures. The method involves dumping material ranging from gravel to small boulders from surface vessels. Gravel is thought to be preferable due to the reduced risk of damage to the pile and associated resuspension and because of the reduced hindrance to commercial trawling. The method is know to smother benthos living in or on the sediments, however, the seabed close to the cutting pile is unlikely to have a diverse benthic community as a result of the presence of the contaminated sediments, gravel dumping of drill cutting piles is therefore considered to have a geographically limited impact on marine organisms, it is also possible that the gravel piles will act as artificial fish attracting reefs.

Insitu Redistribution of Cutting Piles

Oil-based cuttings piles on the seabed can be a source of contamination. Cuttings at the aerobic surface layer of the pile degrade more quickly than those beneath and form a weathered crust; consequently anaerobic degradation within the cutting pile soon ceases. In-situ spreading of the drill cuttings to disperse material across the seabed, increases the oxygenated surface, and is therefore proposed as a means of increasing the rate of biodegradation.

Drill cuttings may simply be spread in a thin layer over the seabed to increase the oxygenated surface layer, or may be blended with the seabed sediment by ploughing or harrowing. A variety of methods could be used to spread the pile, in-situ techniques which act directly on the pile include harrowing and jet propulsion techniques. The first method rakes over the cuttings to flatten it using an anchor plough, drag chain, trawl or rake; the second employs a propeller suspended from a platform or vessel to disperse the cutting by means of hydrodynamic excavation. The only method which has been used with specific intension of spreading cuttings in an attempt to increase biodegradation is trawling by Hamilton Oil Company Ltd, now BHP at Crawford field.

Common Stakeholder Issues

Remediation for the offshore oil industry and for the marine aggregates industry involves a similar community of stakeholders and both are subject to a number of common political, social, economic and technological constraints.

Both operate within the same general principles and aims of UK and international environmental protection policies which regulate the marine environment but within which there is considerable scope for interpretation. This is particularly true in relation to policies to promote biodiversity.

"*Science can measure an impact on the diversity of an area of seabed but whether this impact is deemed to be acceptable or whether a form of protection policy is deemed to be required is a question of value judgement and is not a scientific question*." (CER 1999)

In relation to the North West Hutton decommissioning, it has been recognised that the various stakeholders are likely to have different agendas, with correspondingly variable evaluation criteria. BP have opted for iterative **comparative assessment** as a means of striking a balance between possibly conflicting interests. Such a balance is not simply a matter of scientific determination, but relates to such ill-defined notions as 'public opinion' and 'significance' (or 'importance') (CER 1999).

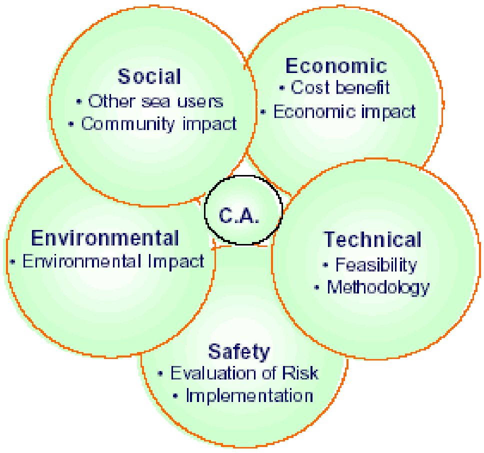


Figure F.3: Function of Comparative Assessment (CA) to resolve conflicting issues in relation to oil rig decommissioning.
(Source: BP Stakeholder Presentation for North West Hutton Decommissioning Project)

Where appropriate **multi-attribute utility evaluation** has been used to make stakeholders' values more explicit, and thereby improve the quality of any multi-criterial debate. (CER after Clemen, 1998). The approach forces assignment of a relative value on different criteria such as ecological impact vs. energy usage. The experience gained with these techniques in the North Hutton decommissioning will provide a valuable source of reference for the marine aggregate industry.

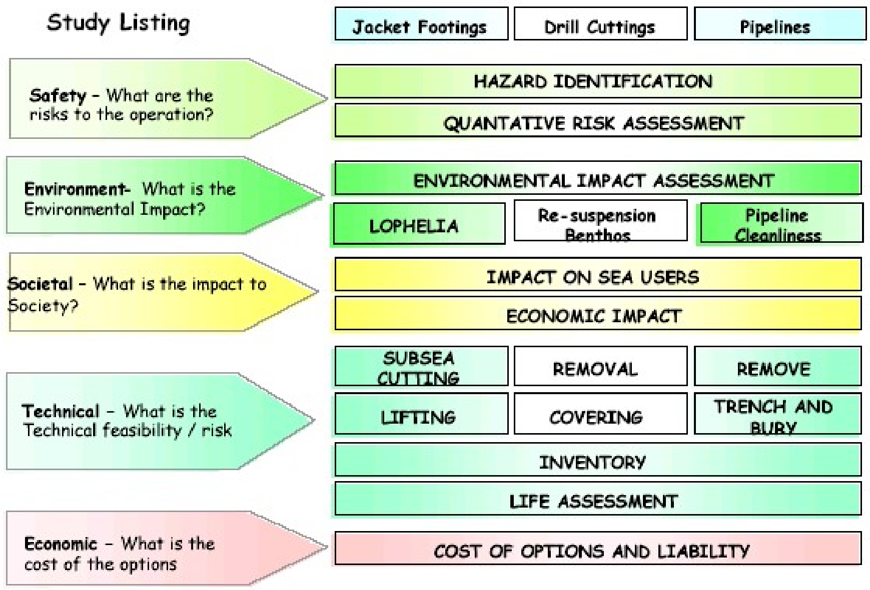


Figure F.4: Summary of issues under consideration in CA process for North West Hutton decommissioning.
(Source: BP Stakeholder Presentation, North West Hutton Decommissioning Project)

3) Example of Topographic Restoration - New York Harbour

This is under consideration as part of the Dredged Material Management Plan[1] for the ports of New York and New Jersey. The DMMP must identify how much material has to be dredged to maintain

[1] The DMMP provides a good example in which ecosystem management has been applied on a regional basis. The basic premise is that the relative combinations of types of habitats, as well as their individual amounts, determine the ecological viability of an area. Habitat restoration efforts within the DMMP seek to target re-establishment of the habitat ratios (to the extent practical in urban areas) present when the area's ecosystem was considered healthy. The approach to restoration balances the needs of the resources in question with coordination among the various interest groups pursuing restoration opportunities

the Federal channel(s) and how that dredged material will be managed in an economically sound and environmentally acceptable manner. The intent of the DMMP is, whenever feasible, to maximise the use of all dredged material as an important resource.

New York Harbour contains several dozen former borrow pits, the largest of which are located in Jamaica and Lower Bays (See Figure F.5). Some of these pits have remained a viable habitat for fish and other estuarine organisms. However, benthic data collected from all pits and analysed while showing significant differences overall, have not identified any particularly productive or unique habitats. The Jamaica Bay pits and West Bank pits were found to be poor environments for most marine life. This is possibly attributable to the accumulation of oxygen demanding sediments, geometry (relatively deep holes with steep sides in a naturally shallow estuary and the lack of hydrodynamic flux resulting in high residence times (particularly the Jamaica Bay pits).

The intention is to fill the pits with category II and III maintenance dredging material (unsuitable for unrestricted ocean disposal) allowing space for a clean sediment cap to isolate the underlying material from organisms living in the water column and the upper portion of the adjacent sediments.

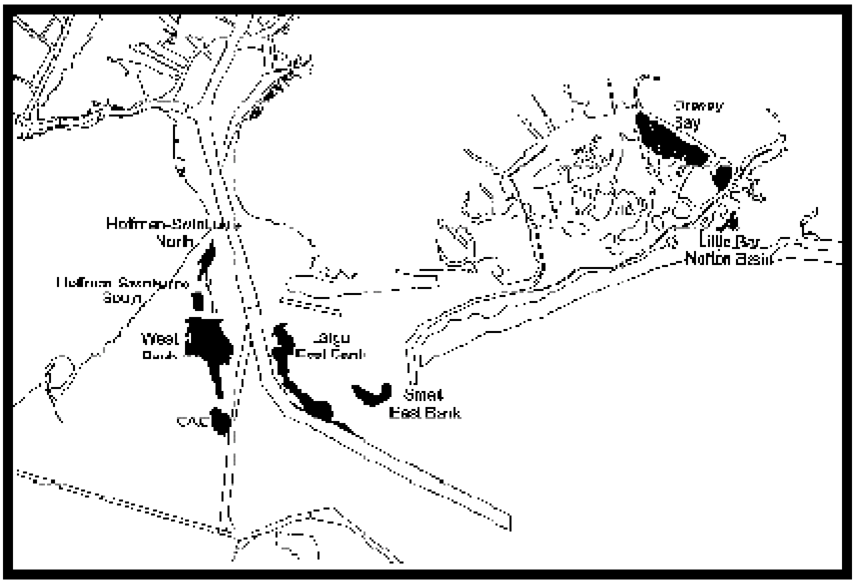


Figure F.5: New York / New Jersey Harbours – former dredging pits, under consideration for topographic restoration. Source: USACE 1999

The nature and contouring of the cap would depend upon the chemical and physical character of the underlying sediments, although would take into consideration habitat requirements for fish / shellfish etc. Internal partitioning and other revisions of pits may prove necessary in certain cases to maximise their safe use. A pre-, interim and post-placement monitoring programme forms part of the plan.

Potential benefits

- Economic disposal option for dredging material**.**
- It is considered that filling of the pits provides an opportunity to restore the sites to conditions more closely representing their natural state, with no associated long-term loss of habitat or benthic communities.

- The underlying rationale for the scheme is to identify a safe and economic disposal method for maintenance dredging material. All other considerations aside, the dredging pits, located close to the navigation channels represent an ideal option minimising the cost of transporting dredged material. Use of the pits potentially represents up to two decades worth of maintenance dredge placement options.

- Potential water quality improvements.

- Based upon hydrodynamic and water quality monitoring of the area, one of the potential benefits of re-contouring specific channels is that it could lead to an overall improvement in water quality.

Potential Impacts

- There will be some loss of habitat for the communities currently occupying the pits.
- Potential impacts to water quality.
- Potential unforeseen hydrodynamic impacts.

The principal concern relates to the potential loss of contaminants from dredged material during placement. This concern is not without grounds since use of a similar approach in Boston Harbour resulted in a failure to effectively cap the infill. It is now recognised that capping commenced too early (within 2 weeks of placement) before the material within the pit had consolidated. The view is that this impact on water quality can be minimised by using tidal currents to confine any dispersal within the pit boundary and allowing between 60 and 120 days for the infill to consolidate.

Topographic restoration of former dredging pits within New York / New Jersey harbours is still awaiting approval which will follow consideration of the final Environmental Impact Statement.

4) Restoration of Dredging Spoil Dump Sites - New York Bight-Dredging Material Disposal Site (US Army Corps of Engineers 1999)

In August 1997, the use of the New York Bight-Dredging Material Disposal Site was terminated. The site was simultaneously designated as the Historical Area Remediation Site (HARS). The HARS is being remediated with suitable dredged material that will not cause significant undesirable effects including through bioaccumulation. This is the first time in US History that dredge material is being used to remediate contaminated areas of the ocean floor.

Since the mid 1800s the site has been used for disposal of a range of material including garbage, city refuse, construction rubble, floatable materials and maintenance dredging spoils. Existing seabed sediments which have the potential to cause adverse effects are being capped with cleaner sediments which meet the criteria of the Ocean Dumping Act. Remediation is a dual process of reducing impacts to an acceptable level and improving the habitat conditions for bottom dwelling organisms.

Based on current projections, remediation of the HARS will require an estimated 40 million cubic metres and will utilise all suitable dredge material for at least the next decade.

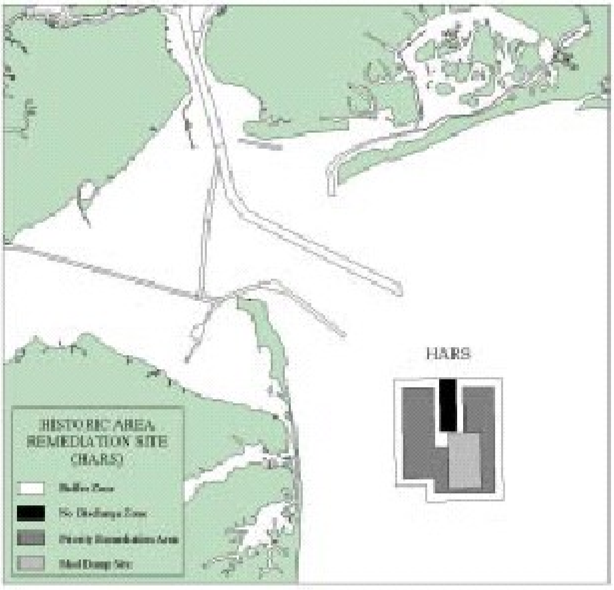


Figure F.6: New York Historical Area Remediation Site (HARS)
[Source USACE New York District]

F.4 Examples of Enhancement of the Seabed

1) Enhancement of Seabed Substrate for Shellfish

Oysters and Clams

The use of sand/clean silt covered with oyster shell as a hard substrate for the settlement of planktonic stage oyster for oyster bed establishment has been done for many years and could also be applied to other target species. For example, clean sand from maintenance dredging could be used to promote soft clam colonisation, by replacing sediment in eroded areas to restore bottom topography and enhance benthic habitat; this could also be modified for hard clams by adjusting the fine/coarse composition of the sediment (US Army Corps of Engineers 1999).

The American Littoral Society 'The Baykeeper' is currently conducting an experiment involving the placement of oyster shell in the Upper Bay of New York Harbour; and the shells are being monitored for oyster larval attachment and growth (US Army Corps of Engineers 1999).

Figure F.7: Depositing Oyster Shell, Raritan Bay, USA
Source:Baykeeper webpage, New Jersey

There is growing realisation that enhancement of this shellfish will also benefit from the restoration of epifaunal reefs. In the Foveaux Strait, New Zealand the commercial densities of oysters occur on bryozoan-dominated reefs, where mortality is probably lower, recruitment higher and growth faster than on other substrates. Heavy fishing with dredges damages and destroys these reefs, which appear to provide the very basis for the existence of this fishery; however restoration may be possible through appropriately returning oyster shell (Booth 2000).

A trial of returned shell with young oyster attached in the Strait has proved successful in establishing a reef and in enhancing oyster density. Therefore enhancement of the oyster fishery and restoration of the beds is likely to involve the spreading of seed oysters and attempts at reef restoration by more focussed distribution of shell. Other species whose growth and survival are associated with intact biogenic reefs are also likely to benefit from these mitigating measures.

Similar mitigating measures are also being investigated in Galveston Bay, Texas (Coastal America Partnership 1996); where over 2 million pounds of eastern oysters are harvested each year. However in recent years oyster production has been in decline due to increased sedimentation over historical oyster beds and a decrease in the amount of suitable reef material for the settlement of juvenile spat. In an attempt to resolve this problem, new oyster reefs have been constructed providing a hard substrate to encourage oyster spat settlement. These reefs are constructed out of pellets made from fly ash, which is produced in large quantities locally.

Scallops

The restoration of the sea bed can also increase the productivity of scallop fisheries, as there is increasing evidence that the nature of the seafloor ultimately determines scallop survival and levels of production. Scallops live on seabeds with a predominant cover of sand and mud, mostly commonly at depths of 10-3m. In Nelson's Tasman and Golden Bay, New Zealand, dredging has altered the nature of the seabed in places and this disturbance is thought to have contributed to declines in natural settlement and spat survival. The restoration of the seabed by returning processed shell is currently under discussion (Booth 2000).

The laying out of old shell and young shellfish onto the seabed may be viewed as having a positive impact, as the shells help to stabilise the seabed and promote re-growth.

Case Study: Elliott Bay, Seattle

A substrate enhancement experimental project has been undertaken at Elliot Bay, Seattle, Washington, with the aim of improving the nearshore marine habitat and enhancing benthic fauna productivity and improving the attributes that support resident and migratory marine and estuarine fish. The aim was to increase diversity of the seabed from the current shallow subtidal sandflats; this was undertaken from barges deploying cobbles, quarry spoil, pea gravel, and oyster shell in separated plots approximately 3-4m^2 in waters between 0.5 and 10m depth. Monitoring of the project is currently being carried out (Duncan and Stark 2000).

Initial results from a five year monitoring programme have been documented in Stark and Cordell (2000). The materials placed have proved effective in providing macroalgae and sessile invertebrate attachment sties. The macroalgae is providing cover for several fish species including sole and rockfish. A variety of invertebrates, including crabs and shrimps are exploiting the cover provided by the substrates and macroalgae.

The pea gravel was specifically deposited to provide habitat for juvenile salmonoid prey. Conclusive statistical evidence cannot yet be compiled to demonstrate the effectiveness of the habitat creation because of the inherent high variability between samples of epibenthic fauna. However, general observations suggest that the overall abundance of epibenthic invertebrates has increased with significant increases in both *Tisbe* spp. and *Dactylopusia vulgaris*, known juvenile salmonoid prey.

The oyster shell habitat was specifically created to attract Dungeness crab. To date, Dungeness crab have not colonised the oyster shell plot, although the plot has provided a complex habitat for other crab species, invertebrates and small fish.

2) Artificial Reefs

Types

Artificial structures represent a spectrum of approaches, high or low technology, manufactured or natural materials and differ in expense, labour and resources. Extreme approaches can range from volunteer efforts to deploy cheap surplus materials e.g. bundles of weighted brushwood, to public and private partnerships that use large fabricated concrete structures (Seaman and Sparague 1991). In addition to providing hard substrate, structures are also used to alter the hydrodynamic regime in order to capture and prevent dispersion of planktonic larvae.

Figure F.9: Tyre Reef, Poole Harbour
Source: Dr. Ken Collins,
Southampton Oceanography Centre.

Studies into the Effectiveness of Artificial Reefs

Crucial to the assessment of the impact of a reef is whether or not it attracts and concentrates marine species rather than increases production. Nelson W. *et al.*(1994) found that fish were recruited to a new artificial structure almost immediately. If a reef merely aggregates species, then construction would not serve a conservation ethic and artificial reefs would be best viewed as fishing gear, like FADs (discussed below). However, many studies have been undertaken to show that artificial reef can not only increase productivity but also biodiversity of an area of seabed (Jensen 1997, Seaman and Sprague 1991, and Jensen, Collins and Lockwood 2000).

Part of the reason for the success of some artificial reefs in supporting high densities of organisms is related to increased habitat complexity, design elements recognised to be most important include physical shape, material composition, surface texture, reef size and dispersion of units (Bohnsack 1991).

A study for the potential of an artificial reef to mitigate for the productivity loss of shallow water habitat was undertaken in the Delaware estuary (Burton *et al.*2002). A reef consisting of 16 pre-fabricated concrete artificial reef structures was constructed in the lower estuary as out-of-kind mitigation to replace a loss of 85 acres of inter-tidal mudflat and 147 acres of subtidal soft-bottom habitat in the upper estuary. Estimates of secondary benthic production using production: biomass ratios compared gains in benthic epifaunal communities to losses of infaunal communities in the upper estuary. Results indicated that the artificial reef produced enhanced benthic secondary production per unit area over the lost habitat, but the total production fell short of replacing the lost production. However, the reef was found to be successful in providing enhanced benthic habitat over areas where no natural reefs occur nearby. Therefore while artificial reefs may not entirely mitigate for the loss of habitat, there is evidence that they can provide enhanced benthic habitat, particularly in areas of low productivity or degraded habitat.

In Canada, artificial reefs specifically for lobster research were built in 1965 from quarry rock, 2-2.5km away from major concentrations of lobsters and monitored over next 8 years. The reef was initially colonised by large specimens, thought to have out grown their burrows and forced to seek new shelter, but by 1973 the size distribution on the artificial reef was similar to that of natural reefs in the area (Scarratt 1973).

In the UK work has been undertaken since 1988 on a reef placed in Poole Bay, which is deployed on a flat sandy seabed and 3km from existing lobster habitat. Within 3 weeks of deployment lobsters were present on the reef and tagging studies have found that the reef is a suitable long term habitat (Jensen and Collins 1996).

Artificial habitat for lobsters has also been considered in the US, with small artificial single and 3-chamber units deployed on the seabed, the number of lobsters found to inhabit these structures were greater than the nearby natural reefs (Sheehy, 1976).

In Israel research has focused on slipper lobsters, which were found to inhabit tyre reefs. These lobsters migrate to deeper waters as temperatures rise but were found to return to the tyre reef over a 3 year period.

For Rock Lobsters the most successful enhancement technique practiced albeit experimentally is the provision of artificial shelters for settling pueruli and young juveniles. In the one trial so far reported for *Panulirus argus* in Florida, simple shelters consisting of pairs of holed bricks placed one on top of another were shown to significantly enhance juvenile survival and abundance (Booth 2000).

A literature review of the attraction and enhancement of finfish by artificial reefs was undertaken by Santos *et al.* (1996), and included 11 artificial structure case studies. The review concluded that artificial reefs are efficient in aggregating fish, that reef attraction is species selective, that fisheries enhancement is more evident as a result of an increase of fish accessibility. The review also highlighted the lack of quantitative reports with objective data and suggestions were made for future studies.

Artificial Reef Structures Designed to Alter the Hydrodynamic Regime

Extensive habitat construction has been undertaken for molluscs in Japan, where marine aquaculture grounds have been created by the deployment of offshore breakwaters, the excavation of channels and the replacement or removal of existing substrates.

The aim has been to cause stagnation and local accumulation of drifting larvae and eggs, thereby preventing attrition and dispersal of juveniles and enhancing their settling opportunity on local grounds. This was achieved by utilising the current shadow zones behind an object placed in a current or the wave-induced circulation around offshore dikes. A good example of this is found south of Cape Masaki in Taro, Japan, where large scale marine farming ground for abalone has been constructed. The aim of the project was to prevent the dispersal of floating eggs, larvae and kelp spores and drifting seaweed by longshore currents, and involved the installation of circulation-inducing structures including breakwaters and submerged dikes. Following the reef construction the area inshore of the submerged dikes developed an algal community inhabited by fast growing abalone and sea-urchins (Fabi and Fiorentini 1996).

The only European model based on Japanese concept is the ‘intensive multi-purpose reef’ deployed off the central Adriatic Coast, Italy; this model combines seabed artificial reefs, with shellfish culture equipment. The aim is to increase the settling opportunity of drifting bivalve larvae and juveniles whose distribution can then extend over the reefs and wider area. This has lead to the deployment of large scale commercial multi-purpose artificial reefs along the Adriatic coast, and allowed the development of new exploitable populations mussels and oysters, in areas of sandy bottom substrate, significant distance from rocky habitats (Fabi and Fiorentini 1996).

In Europe the most commonly farmed shellfish species are mussels, oysters and scallops. Cultivation is undertaken using surface or submerged floating structures (artificial structures), whose aim is to capture drifting larvae by providing suitable substrate and increase the growth rate of the farmed species (Fabi and Fiorentini 1996).

Impact of Artificial Structures

By their nature artificial reefs have the potential to impact on natural processes, both biological and physical, in the marine environment. In Japan huge concrete reefs have been constructed in order to modify marine currents, to produce local upwellings (Ceccaldi 2002). In the past many artificial reef materials have moved, altered, been destroyed or disappeared. The large range of potential physical and biological impacts is detailed by Sheng (2000) and includes the following:

- Slowing of currents
- Bringing sedimentation or scouring
- Increasing turbidity
- Increasing nutrients, algal blooms and changes in water quality through sediment resuspension.
- Cause upwelling
- Destroy nearby natural habitat by disintegrating or moving
- Allow species new to an area to establish
- Increase predation of target species by harbouring predators

Figure F.11: Washed up Tyre Reef

Impacts of an Artificial Reef on Surrounding Habitats

Any new structure placed on an undisturbed seabed is likely to have an impact on the existing ecology, most noticeably by the smothering of the benthic habitat beneath its foot-print. This structure also has the potential to impact the surrounding ecology, by changing the physical process of the area and introducing a new habitat type and associated species into an area where they previously did not exist.

A two year survey of benthic community inhabiting the natural sand-muddy bottom immediately surrounding an artificial reef deployed in the central Adriatic, found significant differences in the abundance, mean species richness and diversity between survey sites adjacent to the reef and the control sites (G. Fabi *et al.* 2002). The reef was located in a depth of 11m, 1.2 miles offshore, far from hard natural substrates and was constructed in 1987 consisting of 29 pyramids made up of five 2m cubic concrete blocks.

Studies undertaken 1975-77 (Davis *et al.*1982), examined the effect of man made structures on natural sand bottom communities in shallow water in Southern California. Here shallow scour effect was found to a distance of 15m around the reefs, but beyond had no effect on the grain size, organic carbon and infauna. However foraging by reef-associated fish significantly altered the epifaunal populations of the sea pen, *Stylatula elongate,* and within 5 months the sea pen was eliminated from distances greater than 200m around the reef. Conversely to this observation, the density of tube building polychaetes Diopatra sp. was enhanced in the immediate vicinity of the reef.

Investigations into the effect of a reef on the surrounding benthos have also been undertaken on a reef in Florida, where four small scale artificial reefs were installed off east coast of Florida in 1987 (Nelson W. *et al.* 1994). Here it was found that fish were recruited to the reefs immediately and the number of fish steadily increased of next 2.5 years. Two years after installation significantly reduced infaunal abundance was observed in the immediate area surrounding the reef (<1m from the reef). This pattern of decreased infaunal abundance near the reef is the opposite to that observed in reef studies on the west coast of the US; where the most conspicuous effect of the reef was the increased abundance of the tube-dwelling worm Diopatra sp. which occurred in close association with the reef modules, increasing the total infaunal density (Ambrose R. and Anderson T. 1990). This reef is located off Southern California and was constructed in 1980 from quarry rock piled in eight piles. The study was undertaken in 1986.

Changes in infaunal abundance on the US east coast did not appear to have been caused by changes in sedimentary characteristics; this suggests that the infaunal reductions are as a result of a combination of direct predation and physical disturbance of the sediment nearest the reefs by the fish using the reef as shelter (Nelson W. *et al.* 1994); whereas on the west coast no evidence was found that foraging by reef-associated fish caused widespread reduction of infaunal densities near the reef (Ambrose R. and Anderson T. 1990). However, evidence was found that the reef altered the grain size distribution of sediments, with coarser sediments close to modules.

Impacts from restoration of crustacean fisheries

- Transfer of unwanted organisms with the transfer of lobsters
- Littering of seafloor where there is widespread installation of artificial shelters and reefs
- Artificial reefs and shelters causing unwanted or unpredictable changes in the ecosystem
- Possible undesirable interactions with other coastal species
- Increased levels of predation by the release species on other key species

3) Fish Aggregating Devices (FADs)

Fish Aggregating Devices (FADs), are objects floated on the surface or suspended in the water column to attract fish by exploiting the natural aggregating behaviour of pelagic fish around floating objects. FADs can be deployed at the surface as a raft or buoy, or at various levels in the water column, depending on the intended target species. FADs are generally anchored to the seabed and can be used alone or suspended above a benthic reef. (Bohnsack *et al.* 1991). FADs are amongst the oldest habitat enhancement practices, in the Philippines and northern Mediterranean surface rafts of bamboo and cork have been employed for centuries (Seaman and Sprague 1991).

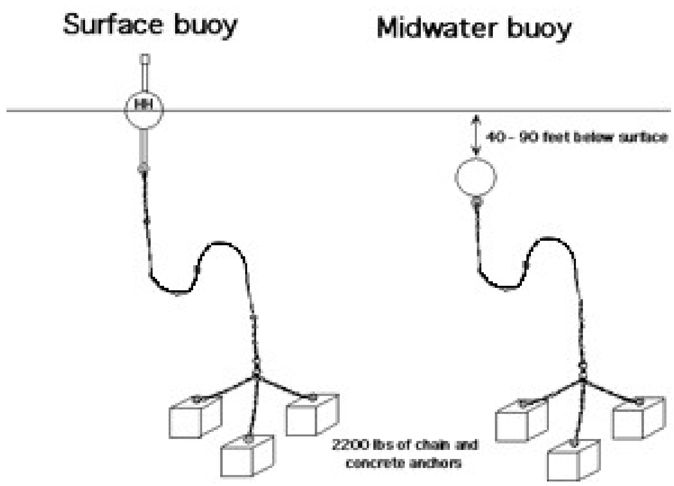


Figure F.13: Two types of Fish Aggregation Device used by the State of Hawaii´

Despite the success of FADs for improving harvests an unresolved question is how fishing at FADs is affecting fish populations. If FADs only aggregate fish and make them easier to catch this could lead to overexploitation, the same argument could apply to artificial reefs especially if they act primarily as fish aggregators.

4) Fertilising the Sea (Booth 2000)

Animal growth is ultimately dependant on primary production and in turn on the availability of dissolved nutrients, particularly nitrate, ammonia, phosphate and silicate. For coastal waters, eutrophication rather than nutrient deficiency is often the issue, there are however instances where phytoplankton growth may deplete nutrient reserves, particularly nitrate in surface waters and lead to levels o f primary production insufficient to support crops. This is mostly likely in enclosed waters under stratified conditions (Gall *et al.* 2000), in such a case the waters could be fertilised to raise the general level of productivity and hence growth of a particular species.

The most significant recent fertilisation experiment is the large Norwegian research programme, Maricult which aims to study the potential of increasing harvests by fertilising both pelagic and benthic ecosystems. In this study small to medium scale fertilisation trials were taking place in landlocked as well as offshore bodies of water (Sakshaug *et al.* 1995); however, there is concern over such large-scale fertilisations. There is also considerable interest in offshore enhancement of primary production; three successful ocean-fertilisation experiments have shown that when limiting trace elements (particularly iron in the Pacific) are added to the sea, there can be greatly increased primary production (Chisholm 2000).

Impacts

There is concern that the volume of fertiliser required to make any difference would be huge, although the possible benefit would be the entrapment of CO_2, the negative impacts include:

- Eutrophication, particularly in enclosed areas, possibly leading to the destruction of the common-owned resources such as fish and shellfish.
- A possible change in population structure of the phytoplankton potentially leading to toxic algal blooms.
- Unpredictable shifts in the ecosystem balance.
- Increased sedimentation.

5) Ecosystem modification

The potential of ecosystem modification has been considered in New Zealand by Booth (2000). Paua (abalone) and Kina (sea urchins) occupy similar subtidal habitats and are frequently found in close association, and at high densities these species seem to exclude each other. It may be possible to rehabilitate or enhance Paua populations by removing Kina (e.g. by fishing) from extensive barrens before introducing Paua to the area. If successful these techniques could be considered for use in areas where Paua are important but threatened through Kina population explosions.

ANNEX G

Generic Risk Assessment for Evaluating the Impacts of Dredging on Fisheries on a Site Specific Basis Developed by CEFAS

Source: A Procedure to Assess the Effects of Dredging on Commercial Fisheries, Final Report, Carlin D. and Rogers S., 2002, CEFAS, Lowestoft.

CEFAS have developed and are currently using a formalised risk assessment process which standardises the data collated for environmental impact assessments and evaluates the impact of each aspect of a dredging operation on the full range of generic fisheries issues. The approach has the merits of achieving greater standardisation and objectivity in data collation while incorporating informed scientific judgement in the assessment process.

The risk assessment involves five stages:

Stage 1 A description of the project, including enough technical information to allow an assessment of its impact on commercial fisheries and fish resources.

Stage 2 The identification of all the impacts of dredging activity which may adversely affect fish populations, such as the direct removal of sediment and increased suspended load. A brief description of the impact is provided and its impact on fish populations.

Stage 3 The identification of the **potential sensitivity** of commercial fisheries and fish resources such as spawning grounds, nurseries etc. to each of these potential impacts; the magnitude of sensitivity is described using the range; very high, high, moderate and low. These classifications will take into consideration spatial and temporal considerations both in relation to the biological resource and the human activity.

Stage 4 The **actual vulnerability** of the fisheries issues to potential impacts is described on a site-specific basis. This balances the theoretical sensitivity of a resource to an impact and the local features of the site and the distribution of the biological resource, to derive and actual site-specific vulnerability, described by the range; very high, high, moderate or low.

Stage 5 An overall assessment of the risk of dredging on fish resources is derived using a matrix which combines the potential sensitivity of each part of the fish population to the each individual impact of the dredging operation and the actual vulnerability at the site.

An example is provided for the St. Catherine’s licence area in Tables G.1 and G.2.

It is suggested that a similar model might be used to assess the impacts of dredging on benthic communities and formalise the procedure for determining when and where remediation at former dredging sites is appropriate.

Aggregate Dredging Licence Application: Evaluation Protocol

Applicant Name:	**SOUTH COAST SHIPPING LTD**		
Licence Name:	**ST CATHERINES**		
Licence Number:	**AREA 407**	**Advice Ref:**	A2.46

1. Temporal and Spatial Scale of the Operation

	Brief Remarks
Benthic Fish Community	Diverse benthic community with abundant structural fauna (incl. Sabellaria) and an important source of food and shelter for crab (edible and spider) cuttlefish, and other resources for bass and bream. Direct uptake of fauna will lead to reduced abundances of these species in the area. Removal rate (1m t per yr) is average.
Fisheries Resources: breeding / spawning grounds	Most important spawning species appear to be bass and lobster. Limited impacts of the scale of the operation on these species.
Fisheries Resources: nursery grounds	Thought to be an important settlement area for crab and lobster.
Fisheries Resources: over-wintering grounds	No evidence that the site is used for overwintering by crab.
Fisheries Resources: migratory routes	Pre-adult female crab, bream, flatfish and pelagics species may pass through the site during seasonal spawning migrations, although the extent of their use of the site is unknown.
Fisheries Resources: direct mortality	Egg-bearing lobster likley to be impacted by direct uptake to moderate extent - no evidence of edible crab spawning at the site.
Commercial Fishery: reduction of income	Unlikely that dredging will seriously affect the alleged 10 bass boats at the site. Some potential impact on the lobster resource - apparently exploited by 4 potting vessels.
Commercial Fishery: displacement of vessels	No evidnece that the site forms important part of local fisheries. Some boats visit the site on a seasonal basis, but no evidence that this is the only site in the region which provides the resources which they target.

2. Method of Aggregate Extraction

	Brief Remarks
Benthic Fish Community	Use of static and mobile dredgers has the potential to change seabed topography and thus adversely affect settlement and distribution of important prey species. Lobster resource in unlikely to be adversely affected by changes in demersal assemblage per se, and bass also likely to find alternative feeding grounds.
Fisheries Resources: breeding / spawning grounds	Limited potential for noise disturbance or impact of changes in seabed topography, to adversely affect spawning fish and shellfish resources.
Fisheries Resources: nursery grounds	Not an important nursery for flatfish, some juvenile crab may be affected to a moderate extend by localised disturbance.
Fisheries Resources: over-wintering grounds	No known edible crab overwintering areas specific to the application area.
Fisheries Resources: migratory routes	No migration routes specific tot he application area.
Commercial Fishery: reduction of income	Potential reduction in income of the fisheries by 10% (p. 8), but this thought to be unlikely, and evidence for consistent fisheries, and accurate fisheries values from the region, is poor.

3. Plume Effects

	Brief Remarks
Benthic Fish Community	A diverse benthic fauna which will be adversely affected by sediment plumes during screening.
Fisheries Resources: breeding / spawning grounds	Some moderate likelihood of impact on lobster resource of sedimentation from plumes at the site and in the immediate vicinity.
Fisheries Resources: nursery grounds	Juvenile shellfish may be afected by plume during screening.
Fisheries Resources: over-wintering grounds	No known crab overwintering areas specific to the application.
Fisheries Resources: migratory routes	Migration routes (if they occur here) are not likely to be impacted by plumes.
Commercial Fishery: reduction of income	Some potential for lobster populations to be deterred from the locality by short-term sediment impacts and longer-term changes to pyhsical substrate.

4. Cumulative Effects

	Brief Remarks
Benthic Fish Community	High density of human activities in region east of the IoW to the north, so some potential for broad-scale changes to assemblages, but very little evidence to supoprt the threat of serious impact. Results of A0903 needed to inform this topic before valid decisions can be made.
Fisheries Resources: breeding / spawning grounds	Combined impacts on shellfisheries in the IoW region have the potential to be significant, but no evidence of stock declines
Fisheries Resources: nursery grounds	Limited potential for combined effects of juvenile shellfish in the IoW region.
Fisheries Resources: over-wintering grounds	No evidence of crab overwintering here.
Fisheries Resources: migratory routes	No evidence of cumulative effects on migration routes in the IoW region.
Commercial Fishery: reduction of income	Potential for the combined effects on shellfish resources of the east IoW licences to adversely affect the populations of crab and lobster, but the supporting science to justify this is Very Weak.
Commercial Fishery: displacement of vessels	Can be argued that vessels displaced fro St Catherines 407 could also be adversely affected by restrictions on access to other nearby grounds from aggregate licences.

Table G.1: Risk Assessment Protocol Test for St Catherine's (Area 407)

Aggregate Dredging Licence Application: Evaluation Protocol

Applicant Name:	**SOUTH COAST SHIPPING LTD**		
Licence Name:	**ST CATHERINES**		
Licence Number:	**AREA 407**	**Advice Ref:**	A2.46

1. Temporal and Spatial Scale of the Operation

	Potential sensitivity		Actual vulnerability: Very High	High	Moderate	Low	
Benthic Fish Community	Very High			1			✓
Fisheries Resources: breeding / spawning grounds	Very High					1	✓
Fisheries Resources: nursery grounds	Very High				1		✓
Fisheries Resources: over-wintering grounds	Very High					1	✓
Fisheries Resources: migratory routes	Moderate					1	✓
Fisheries Resources: direct mortality	Low				1		✓
Commercial Fishery: reduction in income	High					1	✓
Commercial Fishery: displacement of vessels	Low					1	✓

2. Method of Aggregate Extraction

	Potential sensitivity		Actual vulnerability: Very High	High	Moderate	Low	
Benthic Fish Community	High				1		✓
Fisheries Resources: breeding / spawning grounds	High					1	✓
Fisheries Resources: nursery grounds	High					1	✓
Fisheries Resources: over-wintering grounds	High				1		✓
Fisheries Resources: migratory routes	Moderate					1	✓
Commercial Fishery: reduction in income	High					1	✓

3. Plume Effects

	Potential sensitivity		Actual vulnerability: Very High	High	Moderate	Low	
Benthic Fish Community	Very High		1				✓
Fisheries Resources: breeding / spawning grounds	Very High				1		✓
Fisheries Resources: nursery grounds	Moderate				1		✓
Fisheries Resources: over-wintering grounds	Very High					1	✓
Fisheries Resources: migratory routes	Moderate					1	✓
Commercial Fishery: reduction in income	High				1		✓

4. Cumulative Effects

	Potential sensitivity		Actual vulnerability: Very High	High	Moderate	Low	
Benthic Fish Community	High			1			✓
Fisheries Resources: breeding / spawning grounds	Very High				1		✓
Fisheries Resources: nursery grounds	High				1		✓
Fisheries Resources: over-wintering grounds	High					1	✓
Fisheries Resources: migratory routes	Low					1	✓
Commercial Fishery: reduction in income	High					1	✓
Commercial Fishery: displacement of vessels	High				1		✓

Risk Matrix:

	VH	H	M	L
VH	1	1	3	3
H	0	1	5	6
M	0	0	1	3
L	0	0	1	2

Overall Environmental Risk:

High	3
High / Medium	3
Medium	3
Medium / Low	**11**
Low	4
Near Zero	3

Protocol Complete

Table G.2: Risk Assessment Protocol Test for St Catherine's (Area 407) continued…

(Source: A Procedure to Assess the Effects of Dredging on Commercial Fisheries, Final Report, Carlin D. and Rogers S., 2002, CEFAS, Lowestoft.)