The Centre for Environment, Fisheries & Aquaculture Science

A Workshop to Discuss the Feasibility and Practicality of Developing Ecological Quality Objectives for Aggregate Extraction Areas

Held at CEFAS Lowestoft Laboratory on 11 and 12 October 2001

FINAL REPORT

CEFAS Lowestoft Laboratory
Pakefield Road
Suffolk, NR33 0HT

EXECUTIVE SUMMARY

1. A workshop to discuss the feasibility and practicality of developing Ecological Quality Objectives (EcoQOs) and associated qualitative targets designed to protect the marine environment in the vicinity of marine aggregate extraction areas was held on 11th and 12th October 2001 at CEFAS Lowestoft.

2. The background to the workshop was the OSPAR Strategy for the Protection and Conservation of the Ecosystems and Biological Diversity of the Maritime Area and the development of EcoQOs to assist with the implementation of that approach.

3. It was agreed that the principal environmental concerns with the activity of marine aggregate extraction arise from physical impacts at the seabed and the consequential biological responses. Chemical contamination is unlikely to be an issue in most cases due to the very low organic and clay content of commercial aggregate deposits and to the fact that most of these geological deposits would not have been exposed at the surface of the seabed prior to dredging.

4. The workshop recommended 4 overarching EcoQOs for the management of marine aggregate extraction and envisaged that metrics developed for biological and physical impacts would contribute to their assessment:
    - EcoQO 1: To have a proportion (x%) of each habitat that is protected from human activities.
    - EcoQO 2: To ensure that the proportion of habitat and associated communities impacted does not prevent the proper functioning of that system during extraction and allows recovery once dredging ceases.
    - EcoQO 3: Incorporate best practice in dredging operations in order to promote the recovery of impacted ecosystems.
    - EcoQO 4: There should be no impact outside an agreed area of influence; this is the area of primary and secondary impacts. This will be defined and assessed as part of the Environmental Impact Assessment**.**

5. The first two of these objectives could be used directly for other human activities impacting the seabed. The latter two could probably be adapted to do so.

6. EcoQOs which are in the future developed for a variety of human activities may overlap in their areas of application, for example in the case of a coastal locality which is subject to multiple uses. In such circumstances, the potential for conflicting and inconsistent EcoQOs must, at an early stage, be addressed by appropriate co-ordination of groups responsible for their development.

7. Lists of physical and biological metrics appropriate for use in assessing potential environmental impacts of marine aggregate extraction were developed and agreed by the workshop. A number of these metrics would be applicable to other human activities impacting the seabed.

8. Specification of the overarching EcoQOs, the metrics to support them and the monitoring programmes to provide information about the status and trends in the EcoQOs and metrics will require further development. However, data exist for a number of variables presently targeted in a monitoring context, so that reference levels and degrees of acceptable change can be derived and then tested for their utility at least for some of the metrics discussed.

9. It was emphasised that consistent time-series information at impacted and reference stations was essential for retrospective testing of the utility and robustness of potential EcoQOs, prior to their application in routine environmental management.

10. The key research requirement identified was the facilitation of cost-effective mapping of the spatial extent and integrity of seabed biotopes. This will enable the identification of those considered to be the most important in terms of ecosystem structure and function, and therefore most in need of protection. This will also have value in determining 'baseline' (pre-dredging) status, and in subsequent monitoring of changes during and after the event of marine aggregate dredging. Such mapping will also be highly relevant to other human activities.

CONTENTS

1.	**INTRODUCTION**		7
	Background to the Project		7
	Scope of the Workshop		8
	The Workshop		9
2.	**WORKSHOP PAPERS**		**11**
	Simpson, T	Regulation of marine aggregate extraction in England	13
	Emmerson, R	OSPAR developments on Ecological Quality Objectives	15
	Pearson, T H	The role of science in underpinning the development of summary measures of environmental impact	19
	Elliott, M	Application and applicability of Ecological Quality Objectives and Standards for Dredging and Aggregate Extraction	31
	Rees, H L	Practical Considerations relating to the derivation of Ecological Quality Objectives for marine aggregate extraction activities	41
	Boyd, S E and Rees, H L	Towards an understanding of the impacts of marine aggregate extraction on the seabed as a basis for developing an EcoQO approach for management	47
	Rogers, S I and Nicholson, M D	Applying the EcoQO concept to potential fisheries impacts of marine aggregate extraction: Monitoring the outcome	59
3.	**WORKING GROUP OUTCOMES**		**63**
	Working Group 1.	Develop EcoQOs (analogous to EQOs) taking account practicality, feasibility, regulatory interests, conservation and fisheries interests.	65
	Working Group 2.	Define measures appropriate for use as environmental indicators, their state of development, practicality and feasibility: Sub-group: Biological.	69
	Working Group 3.	Define measures appropriate for use as environmental indicators, their state of development, practicality and feasibility: Sub-group: Physical.	74
4.	**CONCLUSIONS**		80
5.	**APPENDICES**		82
	Appendix 1. Participants		84
	Appendix 2. Workshop Programme		88
	Appendix 3. Further Reading		92

1. INTRODUCTION

Background to the Project

The UK is a contracting party to the OSPAR Convention for the protection of the marine environment of the North-East Atlantic. The OSPAR Commission has adopted a strategy to protect and conserve ecosystems and the biological diversity of the maritime area which are, or could be, affected as a result of human activities, and to restore where practicable, marine areas which have been adversely affected. The strategy identifies sand and gravel extraction as one of a list of human activities, for which an assessment of the actual or potential adverse effects on species and habitats or on ecological processes is required.

Concurrent with these assessments, OSPAR is elaborating techniques for the development of Ecological Quality Objectives (EcoQOs) to contribute to the development of an ecosystem approach to environment management. In 1999, a joint OSPAR/North Sea Conference workshop in Scheveningen, Netherlands identified a number of the general issues for which EcoQOs for the North Sea should be developed (Lanters *et al*., 1999).

The 1999 workshop approved the following definitions of Ecological Quality and Ecological Quality Objective:

> ***Ecological Quality (EcoQ):*** Ecological quality of the surface water is an overall expression of the structure and function of the aquatic systems, taking into account the biological community and natural physiographic, geographic and climate factors as well as physical and chemical conditions including those resulting from human activities.
>
> ***Ecological Quality Objective (EcoQO): EcoQO*** is the desired level of EcoQ relative to the reference level.
>
> ***EcoQO reference level*** has been defined as the level of EcoQO where the anthropogenic influence on the ecological system is minimal.

From the Scheveningen workshop discussions, it was evident that while qualitative targets were not ruled out, efforts towards the identification of quantitative targets will be the preferred outcome for the practical implementation of the EcoQ/EcoQO approach. However, the operational implications of the variety of terms currently employed, and the environmental targets, require further clarification. For example, the Scheveningen Workshop states that 'other terms such as environmental quality standards…can be seen as operational elements or tools used in the context of EcoQOs'. The proposed EcoQ/EcoQO approach is, in a number of respects, comparable with the UK Environmental Quality Objective/Environmental Quality Standard (EQO/EQS) approach, albeit pitched at a higher (ecosystem) level, and with a much stronger emphasis on the employment of measures of biological status. EQOs and associated EQSs for use as compliance measures, have a long history of application for aquatic environmental management in the UK, principally in the control of waste discharges.

The ICES Study Group on Ecosystem Assessment and Monitoring (SGEAM) was set up in 2000 to reflect on the scientific framework for an ecosystem approach for the sustainable use and protection of the marine environment, including living marine resources, and to review the methodology and proposals for Ecological Quality Objectives for the North Sea (ICES, 2001a). The ICES Working Group on the Ecosystem Effects of Fishing Activities (WGECO), reviewed specific recommendations for EcoQOs for sea birds and marine mammals, and provided general recommendations on the future scope and direction of an ecosystem approach to marine environmental management (ICES, 2001b).

The marine aggregate industry have expressed support for the principle of developing quality standards for use in monitoring marine aggregate extraction areas in order to judge the effectiveness of licence conditions, which may also include mitigation measures. Properly formulated, such standards should help to simplify discussions concerning the nature, extent and significance of dredging-induced changes. Of particular relevance in this respect is the recent UK experience in the development of EQO/EQS for the management of sewage sludge and dredged material disposal at sea, arising from the work of the MPMMG's Group Co-ordinating Sea Disposal Monitoring (Anon, 1996 and Rees and Pearson, 1992). However, the amount of information on, and hence the degree of understanding about the effects of, marine aggregate extraction is less well developed than that for the disposal of those materials and we lack an accepted empirical model describing the impacts e.g. analogous to that of Pearson and Rosenberg (1978) in the case of the response of the benthic fauna to organic enrichment.

Under impending legislation to regulate the activity of marine aggregate extraction in the UK, Quality standards could guide the derivation of appropriate Dredging Permission conditions on a case-by-case basis and could be used to assess compliance with those conditions. They would also guide the development of appropriate monitoring programmes by applicants/consultants during the Environmental Impact Assessment process and would provide a more transparent means of determining the acceptability or otherwise of dredging-induced impacts.

Scope of the Workshop

The present workshop was the main component of a research project commissioned by the Department of the Environment, Transport and the Regions, Marine, Land and Liability Division (now the Department for Environment, Food and Rural Affairs -Marine and Waterways Division - DEFRA-MWD) to examine the need for, and feasibility and practicality of, developing Ecological Quality Objectives designed to protect the marine environment in the vicinity of marine aggregate extraction areas. To date, the OSPAR approach to Ecosystem Management and EcoQ/EcoQOs had not generally included their application at the level of individual human activities. However, it is likely that such an approach would be needed eventually to contribute to ecosystem management of the marine environment. This workshop was designed as a test case for the development of EcoQ/EcoQOs for human activities listed in the OSPAR Strategy for Annex V that will be particularly useful in the context of OSPAR's future consideration of marine aggregate extraction. The selection of this activity as a test case was not on account of any particular emphasis on its environmental impacts compared with other human activities. It is also particularly appropriate in a UK context because of the relatively large amounts of aggregate extracted compared with most other OSPAR countries and the imminent enactment of controlling legislation. The research project also has particular relevance to current UK initiatives to develop performance indicators for the management of the marine environment.

It is generally accepted that the principal environmental concerns with the activity of marine aggregate extraction arise from physical impacts at the seabed and the consequential biological responses. Chemical contamination is unlikely to be an issue in most cases due to the very low organic and clay content of commercial aggregate deposits and to the fact that most of these geological deposits would not have been exposed at the surface of the seabed prior to dredging.

It should be noted that the primary focus of the workshop was on the environmental impacts at and near to marine aggregate extraction sites and considerations did not extend to the potential for more distant impacts on coastal erosion.

The Workshop

A Steering Group was established to develop the programme for the workshop and to identify the appropriate attendees. The Steering Group consisted of Dr Richard Emmerson (DEFRA), Dr Tony Murray (Crown Estate), Mr Mark Russell (BMAPA) and Dr Chris Vivian (CEFAS). The invited experts who were able to attend the workshop are listed in Appendix 1. The experts included policy makers, regulators, industry representatives, scientists and NGOs. The workshop programme, the workshop papers and supporting information was circulated to delegates prior to the workshop to inform them of the purpose of the workshop – see Appendices 2 and 3. The workshop was held at the CEFAS Lowestoft Laboratory on the 11th and 12th October 2001.

References

ANON., 1996. Monitoring and Assessment of the Marine Benthos at UK Dredged Material Disposal Sites (1996). Department of Agriculture and Fisheries for Scotland, Report Number 21, 35pp.

ICES, 2001a. Report of the Study Group on Ecosystem Assessment and Monitoring. ICES CM 2001/E: 09, 90 pp.

ICES, 2001b. Report of the Advisory Committee on Ecosystems, Copenhagen, ICES Cooperative Research Report No. 249, 94pp.

LANTERS, R.L.P., SKJOLDAL, H.R. AND NOJI, T.T., 1999. Ecological Quality Objectives for the North Sea. Basic Document for the Workshop on EcoQOs for the North Sea, Scheveningen, The Netherlands, RIKZ Report 99.015.

PEARSON ,T. H. AND ROSENBERG, R., 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Oceanography and Marine Biology Annual Review, *16*: 229-311.

REES, H.L. AND PEARSON, T.H., 1992. An Approach to the Setting of Environmental Quality Standards at Marine Waste Disposal Sites, Marine Environmental Quality Committee, ICES CM 1992/E: 33.

2. WORKSHOP PAPERS

Regulation of marine aggregate extraction in England

Dr Tom Simpson
Minerals and Waste Planning Division,
Department of Transport, Local Government and the Regions
E-mail: Tom_Simpson@dtlr.gsi.gov.uk

Sand and gravel extracted from the seabed makes an important contribution to England's demand for construction aggregate materials. It is particularly important in London and the South East where it accounts for almost a third of the total regional demand for sand and gravel. Using marine sand and gravel reduces the pressure to work minerals on land where it is becoming increasingly difficult to find resources that are not constrained because of their agricultural, environmental or development value.

The Government wish to see its continued use, at a rate which can satisfy market demand, to the extent that this remains consistent with the principles of sustainable development. It is important, therefore, that the marine and coastal environment is protected from significant harm and that fisheries and other legitimate uses of the sea are not unacceptably affected by aggregate dredging activities.

The extraction of sand and gravel from seabed is currently controlled through a non-statutory "Government View" (GV) procedure, which was introduced in 1968 to ensure that the impacts of dredging on the marine environment were considered before a license was issued. The Crown Estate, as owners of most of the seabed around England and Wales, will only issue a dredging licence if the Government, having considered all the relevant information, give a favourable GV. Since 1989 the applicant has been required to undertake an Environmental Impact Assessment (EIA), including a Coastal Impact Study, to support their application.

A statutory system for regulating marine aggregate extraction will be introduced shortly through the Environmental Impact Assessment and Habitats (Extraction of Minerals by Marine Dredging) Regulations. These Regulations will also transpose into UK law the requirements of the European Community Directives on Environmental Impact Assessment (the 'EIA Directive') and the protection of natural habitats of international importance (the 'Habitats Directive') with respect to the extraction of minerals by dredging.

The Regulations will define the Secretary of State for Transport, Local Government and the Regions as the regulator for the control of marine dredging for minerals within English waters. He will be responsible for determining applications for permission to dredge for minerals and for making all other decisions on dredging matters that are required under the Regulations. He will have powers to impose legally enforceable conditions on the dredging permissions, which will be used to reduce or remove potential adverse effects on the environment and require specified monitoring of the impacts of dredging. Procedural guidance on the operation of the Regulations will be published at the same time as the Regulations come into force in Marine Minerals Guidance Note 1 (MMG1).

The Government is also developing a new policy framework to accompany the Regulations, which will seek to ensure that extraction takes place in a way that is consistent with the Government's approach to protecting the environment and achieving sustainable development. A public consultation exercise was undertaken earlier this year on the draft framework (draft MMG2).

OSPAR developments on Ecological Quality Objectives

Dr Richard Emmerson
Marine and Waterways Division
Department for Environment, Food and Rural Affairs
E-mail: Richard.Emmerson@defra.gsi.gov.uk

Within the framework of OSPAR and the North Sea Conferences, the UK is working with other contracting parties to develop a more ecologically orientated management of the seas centred on an ecosystem-based approach. Such an approach requires that management decisions be based upon knowledge about the consequences for the ecosystem as a whole. The paper described the development of Ecological Quality Objectives (EcoQOs) to contribute towards these aims and consider the applicability of this approach to management of specific human activities, for which this workshop is using aggregate extraction as a test case.

1. DEFRA stimulus for the workshop

- Objective 1 of the new department:
  "To protect and improve the environment and conserve and enhance biodiversity, and to integrate these policies across Government and internationally."
- Objective 6 of the new department:
  "To promote more sustainable management and use of natural resources (e.g. energy, water, fisheries, forests etc), in the UK and internationally."

2. To support an emphasis on Marine Conservation, Government is currently preparing a Marine Stewardship Report which will describe the UK's development of an approach to management of the marine environment, based upon the following principles:

- Sustainable exploitation of marine resources;
- Environmental protection;
- Ecosystem approach;
- Greater integration; and
- Precautionary principle.

3. The international context for policy on the marine environment is provided by the UK's commitment to work within the framework of the Oslo Paris Convention (OSPAR Convention) for the Protection of the Marine Environment of the North-East Atlantic which has agreed strategies on the following issues:

- Hazardous substances;
- Eutrophication;
- Radioactive substances;
- Oil and gas installations; and
- Protection and conservation of ecosystems and biodiversity.

4. The OSPAR Strategy of particular interest for this workshop is the Strategy on the Protection and Conservation of Ecosystems and Biodiversity that has the following objective:

> "…to protect and conserve the ecosystems and the biological diversity of the maritime areas which are, or could be, affected as a result of human activities, and to restore, where practicable, marine areas which have been adversely affected…"

5. The strategy identifies a number of human activities that need to be assessed for their impacts on the marine environment, including:
   - Sand and gravel extraction;
   - Dredging for navigation;
   - Placement of oil and gas structures;
   - Placement of artificial islands, reefs, installations and structures;
   - Introduction of GMOs; and
   - Land reclamation.

6. Annex V to the OSPAR Convention on the protection and conservation of ecosystems and biodiversity of the OSPAR Convention states that:

> "…in drawing up programmes for the control of human activities…..to aim for the application of an integrated ecosystem approach."

7. This commitment to an ecosystem orientated approach to management was also made at the Intermediate Ministerial Meeting on the Integration of Fisheries and Environment issues of the North Sea Conference 1997 (IMM 1997), where:

> "Further integration of fisheries management and environmental protection, conservation and management measures drawing upon the development and application of an ecosystem approach."

8. The IMM 1997 identified an ecosystem approach focusing on the critical ecological processes, the ecosystem interactions and the chemical, physical and biological environment. The North Sea Conference (NSC) ecosystem approach was further elaborated by a Workshop held in Oslo 1998 to include the following elements:
   - Integrated management;
   - Clear objectives both operational and general;
   - Better use of scientific information;
   - Focused research on the ecosystem including climatic, biological and human driving forces of ecosystem variability;
   - Improved integrated monitoring;
   - Integrated assessment by fisheries, environment and socio-economic experts; and
   - Involvement of stakeholders, scientists, managers and politicians at different stages of the decision making process.

9. The Oslo Workshop identified the need for objectives to fulfil the aims set out for an ecosystem approach:
   - "…a set of ecological quality objectives (EcoQOs) need to be developed and agreed upon by the parties involved i.e. scientists, policy makers and stakeholders."

- "…objectives need to be at the general level, as overall or integrated objectives, and at the specific level."

10. In order to take this initiative forward there have been a number of meetings to discuss the development of EcoQOs for the North Sea in particular.

- NSTF - 1992 Bristol – 1993 Geilo
- ASMO – 1995 Ulvik
- OSPAR/NSC – 1999 Scheveningen

The Netherlands and Norway are leading the development of EcoQOs under the North Sea Conference.

11. This process had (at the time of the workshop) established the following EcoQ/EcoQO definitions:

Ecological Quality (EcoQ)

Ecological Quality of the surface water is an expression of the structure and function of the ecosystem, taking into account the biological community and natural physiographic, geographic and climatic factors as well as physical and chemical conditions including those resulting from human activities.

Ecological Quality Objective (EcoQO)

EcoQO is the desired level of EcoQ relative to a reference level.

EcoQO reference level has been defined as the level of EcoQ where the anthropogenic influence on the ecological system is minimal.

12. Ten general issues have been identified under the NSC for the development of EcoQOs:

1. Reference points for commercial fish species;
2. Threatened and declining species;
3. Sea mammals;
4. Sea birds;
5. Fish communities;
6. Benthic communities;
7. Plankton communities;
8. Habitats;
9. Nutrient budgets and production; and
10. Oxygen consumption.

13. These 10 issues are seen as an integrated set and a 2-track approach has been suggested, (i) ecosystem and crucial processes, and (ii) human activities which relate to management objectives.

14. To contribute to work on EcoQOs under the OSPAR Strategy to Combat Eutrophication and to stimulate UK thinking on EcoQOs and associated indicators a project to - consider the development of EcoQOs in relation to eutrophication was taken forward at a DETR workshop held in Brighton in 2000.

15. The achievements of the Brighton workshop were:

- General EcoQO statements;

- Classification of water/seabed types;
- Definition of indicators; and
- Definition of problem and acceptance thresholds.

16. A general EcoQO for eutrophication was proposed

There should be no substantial change in phytoplankton communities from the reference level as a result of anthropogenic inputs of nutrients, as indicated by:

i) biomass;
ii) production;
iii) species succession and composition; and
iv) duration of blooms.

17. This work complements and will feed into a wider UK project to develop Performance Indicators to inform progress against the wide range of management objectives relevant to the marine environment. The need to develop performance indicators for and the following issues has been identified:

- Exogenic unmanaged pressures;
- Hazardous substances;
- Radioactivity;
- Offshore industry;
- Bathing waters;
- Shellfish;
- Protected areas;
- Species;
- Dredging, including aggregate extraction;
- Mariculture;
- Shipping; and
- Litter.

Rationale for this Workshop

18. A number of motivating factors and responsibilities has lead to the organisation of this workshop:

- International and national policy objectives;
- Research – aggregate extraction is a test case of a human activity due to the large amount of research data available;
- The work on an ecosystem-based approach to environment and fisheries management in OSPAR/NSC framework;
- The UK project on performance indicators.

19. The desired outputs include:

- Assessment of practicality;
- Develop methodology;
- Identify research needs; and
- Stimulate thinking of EcoQOs.

The role of science in underpinning the development of summary measures of environmental impacts

T. H. Pearson
Seas Ltd, c/o D.M.L.
Oban Argyll, PA34 4AD, UK ; and
Akvaplan-niva AS
Polar Environment Centre, 9296 Tromsø, Norway,
E-mails: seas@dml.ac.uk; tom.pearson@akvaplan.niva.no

Abstract

The scientific basis for setting environmental quality standards (EQSs) for the response of marine ecosystems to anthropogenic disturbances is briefly reviewed. The development of standards based on benthic community response to carbon enrichment and contamination is considered. Initial attempts to devise response indices based on univariate population parameters are discussed and it is suggested that these have been only partially successful and, in some situations, may be at best unreliable and at worst actually misleading. The development and utility of the Infaunal Trophic Index, based on characterising samples on the basis of feeding habits, is examined critically and some serious flaws in its application are discussed. The recent development of response indices based on multivariate ordination techniques is considered and it is suggested that these present a more objective and powerful means of quantifying faunal response to disturbance.

It is recommended that such indices should be should be adapted to disturbance gradients in British coastal waters and used to provide a suitable biological data input to models currently being developed to predict the impact of anthropogenic disturbance to marine benthic communities.

Introduction

Setting standards for ecosystem response to anthropogenic disturbance requires the establishment of reliable and standardised methods for the estimation of the temporal and spatial scale of such response. Such methods should be accurate, objective and cost-effective and flexible enough to be applicable to a wide range of those marine ecosystems impacted by human activities. Experience accumulated over the past thirty years of marine environmental impact assessments has demonstrated that in most areas and circumstances one of the most responsive and easily addressed elements of the marine ecosystem following an imposed disturbance is the benthic macrofauna (e.g. Rees *et al*., 1980). As a result most marine monitoring and assessment studies have been based on recording changes in the distribution, density and community structure of macrobenthic organisms. However the analytical and interpretational difficulties in summarising those multivariate changes and associating them objectively with changes in appropriate environmental variables are considerable (Henderson & Ross, 1995). Initial attempts to establish response standards were focussed on univariate measures but more recently attention has turned to the use of multivariate analyses. The comparative utility of these approaches is considered here.

Establishment and application of EQSs

In order to set an EQS a reliable means of quantifying deviation from an accepted norm must be established. Thus environmental and biological conditions are recorded in reference areas remote from the disturbed areas and compared with conditions in areas subjected to disturbance. The physical and chemical conditions in sediments and overlying water column are relatively easily assessed by recording the values of key univariate parameters, e.g. oxygen, salinity, redox, grain size etc. and comparing value differences between disturbed and undisturbed areas. Biological conditions on the other hand are assessed on the basis of contrasting community structure and/or functions and population densities and/or biomass. Setting standards based on variation in these factors is complicated because of high levels of natural variability and the intrinsic complexity of structural and functional inter-relationships in such communities. Following reviews of monitoring procedures (Rees *et al.,* 1990), initiated by the Co-ordinating Group on Monitoring Sewage Sludge Disposal Sites (GMSD), EQSs based on variation in the three primary population statistics (taxon richness, T, total abundance, A and total biomass, B) were proposed (Rees & Pearson, 1992). These were subsequently adopted by the UK Environment Departments as part of the standard methodology for application throughout the UK. At the same time the use of the Infaunal Trophic Index (ITI) (see below) was recommended for assessing the normality of reference areas when compared with disturbed sites.

Use of indices and models

Many diversity indices have been devised to summarise population variance within and between biological communities. These are most frequently formulated from species richness and abundance counts but may also be based on other community measures, e.g. comparative biomass or functional attributes. They are designed to combine differing aspects of community structure into a single integer or expression in order to enhance comparisons between and within complex population groups. Such indices have, therefore, a particular value in modelling studies. It must be recognised, however, that they all reduce the range of biological variance being expressed and, depending on the exact formulation of any particular index, bias the outcome of ensuing comparisons. Nevertheless the need to establish clear and unequivocal EQSs and to devise models to predict benthic community response to environmental impacts, necessitates the use of such indices. In view of this, the choice of an index on which to base a biological quality standard must be made carefully and its subsequent application examined very critically to ensure that any inherent bias does not invalidate the eventual comparative studies on which management decisions will be made. With this in mind the relative utility of the ITI and some other commonly used diversity measures will be examined in some recent case studies where they have been applied as standard measures.

A critique of the ITI

The use of the ITI index was recommended by the Comprehensive Studies Task Team of GCSDM (CSTT, 1994) for use in the assessment of the impact of sewage discharges on benthic fauna. Its utility was based on studies undertaken in California (Mearns & Word, 1982) and the index developed there by Word (1979) was adapted to use in UK conditions by WRc plc (1992). It relies on the assessment of changes in the feeding (trophic) mode of the dominant benthic organisms in an area subjected to increasing organic enrichment. It has long been known that increasing enrichment favours initially surface deposit feeding organisms and eventually sub-surface deposit feeding organisms at the expense of suspension feeders (Pearson & Rosenberg, 1978). If the relative proportions of these groups present in a benthic community are calculated and compared the relative degree of carbon enrichment in that community can be assessed. The Infaunal Trophic Index (ITI) provides a means of calculating these relative proportions and results in a single figure index for any sample or series of samples. However, there are serious difficulties in defining such an index and in applying it. These difficulties arise partly from how trophic groups are recognised and defined and from the poor level of knowledge on the actual feeding habits of the majority of the organisms in benthic communities. The present ITI defines four major trophic groups, namely detrital feeders, interface deposit feeders, deposit feeders and specialised environment feeders. Each of these are further subdivided into 4-5 sub-groups defined on various modes of food particle capture. Unfortunately there are considerable overlaps between detrital feeding and deposit feeding as defined originally by Word (1979) and the fourth group of specialised environment feeders embraces a wide range of disparate feeding modes including carnivory and sub-surface deposit feeding. This overlap of functions within the groups defined inevitably results in a blurring of any distinction between the groups when summed over a diverse community. If, in addition, the actual feeding mode of an organism is unknown, debatable, or can be switched under varying rates of carbon input (a common phenomenon) then the possibility arises of serious flaws and misinterpretations of the results of community change in response to carbon enrichment. An additional ambiguity in Word's trophic classification that was perpetuated in the WRc version was the absence of any specific class for carnivores.

These can be a numerous group in polluted sediments but obligate carnivorous taxa, e.g. the polychaete *Glycera* and Nemertean worms, have been variously attributed to groups 2, 3, and 4 by different workers in past studies. It should be noted that the appendix to the WRc report evaluating the index listed nearly 200 species from the UK benthos and made at least 80 suspect or arguable attributions to one or other of the four major trophic groups. Anyone using that list as a basis for calculating the ITI would have made serious errors of interpretation. The original list has been updated as new information on the ecology and behaviour of benthic organisms becomes available. Unfortunately detailed information on many of the organisms sampled is unavailable and in many cases trophic attributions have been made by what could be termed 'sympathetic morphology', i.e. that similar shapes imply similar habits. This has lead to numerous miss-attributions, some of which have subsequently been corrected following new autecological studies, but leaving others uncorrected that may unpredictably bias the final calculation of the indices. Since there is no officially recognised corrected list, it is probable that individuals and groups are making different attributions.

Problems also arise in attributing a trophic category to a higher taxonomic group comprising a number of unidentified lower taxa. An apposite example of this is the categorisation of nematode worms. This large class of organisms includes species with every type of trophic habit. But, because of a general lack of appropriate taxonomic expertise in this group, they are not normally distinguished in benthic analyses. The present lists categorise the entire group as carnivores (entered as trophic group 4 in the ITI calculation) when, in fact, it would be more appropriate to split the population count equally between all four trophic groups.

A further problem is that the ITI index was developed and validated on evidence collected almost entirely from fine sediment areas. The fauna in such areas differs both structurally and functionally from that in coarse sediments. The ratios between the various trophic groups, the enumeration of which form the basis of the index, differ widely in coarse sediments where suspensivores and carnivores greatly outnumber detritivores. This suggests that the ITI levels quoted in the original report and in later official monitoring guidelines and protocols as being normal (60-100), may be misleading in the context of coarse sediment habitats.

In view of these perceived difficulties in calculating and applying the ITI index it's reliability and accuracy in representing faunal response to environmental impact can be called in to question. To illustrate it's relative utility in monitoring biological response two case studies are presented.

Case study 1: a comparison of values for various diversity indices from monitoring stations around sewage outfalls

In the course of evaluating the impact of effluent discharges from sewer outfalls in various areas of the west of Scotland grids of sampling stations were established around each outfall. Sediment and water column conditions were assessed at these stations in accordance with the recommendations made by the Marine Pollution Monitoring Management Group (MPMMG) in 'Comprehensive Studies for the Purposes of Article 6 of DIR 91/271 EEC, the Urban Waste Water Directive' (CSTT, 1994). The distribution of the benthos in the sediments was assessed and analysed using a range of univariate and multivariate techniques including calculation of the ITI and a number of other standard diversity indices. Figures 1 and 2 illustrate the field results from the survey around an outfall discharging into the entrance to a small sea loch. These showed ITI values ranging from 35 to 91, with the lowest values found at the stations near the outfall (Stations 1 - 6) and the highest values at stations in the coastal area outside the loch 3-5 km from the outfall (Stations 13, 16 and 17). The stations in the inner bay showed intermediate ITI values. These field values for the ITI index conform generally with the interpretations of the data made on the basis of the results of the standard multi- and univariate statistical analyses and a consideration of the distribution of species indicative of faunal enrichment. These indicated that the stations in the inner part of the bay were as, or more, enriched than those close to the outfall. The ITI indices did not fully support these conclusions, suggesting that the index is not particularly sensitive to faunal changes brought about by intermediate levels of carbon enrichment.

Case study 2: analysis of monitoring survey data from the vicinity of fish cages in a Scottish sea-loch.

A survey of the conditions in the water column and sediments at and in the vicinity of fish farm cage groups is required by the regulatory authorities in Scotland (SEPA, the Scottish Environmental Protection Agency). Should conditions beyond a 25 m allowable zone of effect (AZE) differ substantially from those found at reference stations then action is taken to reduce the permitted production tonnage to a level that will allow recovery. EQSs against which to judge the recorded conditions are set, based on a suite of physical, chemical and biological factors. Amongst these considerable emphasis is put on the ITI values calculated for each station. The SEPA guidelines propose that a biological standard for marine cage farming should be that the ITI value at the 25 m station should not be more that 20 % lower than at the reference station. In order to assess the reliability of the ITI in relation to other recommended measures of biological response a data summary from a fish cage survey at a farm on the west coast of Scotland is presented in Table 1. This lists the dominant taxa present, population statistics and sedimentary characteristics recorded from stations at 0, 25 and 50 m distance from the cages in the direction of the residual current and at two reference stations sited beyond the influence of cage effects at 500 and 800 m from the cage group.

The fauna at the station adjacent to the cages was severely degraded, with only seven taxa recorded. The most numerous of these, *Mytilus*, originated, most probably, from the cage structure rather than the sediments. The latter were sulphidic and had highly negative redox potentials. At the 25 m station the sediments were also sulphidic with negative, albeit slightly higher, Eh values. There the populations were considerably larger but dominated by nematodes and small detritivorous polychaetes indicative of highly enriched conditions. The most numerous taxa at the 50 m station were the sub-surface detritivorous polychaete *Mediomastus* and the small lucinacean bivalve *Thyasisra*, both of which are associated with moderately enriched areas. The total number of taxa found at 25 and 50 m were only half the number found at the 500 m reference station and the H' diversity values were low. The ITI were lowest (< 20) at the 25 m station and intermediate in value (≠ 50) at 0 and 50 m. Three differing ITI values are presented for each station, each based on a slightly different way of selecting the data to be included in the calculation. ITI G1 and ITI G2a differ in whether or not indicator species with shared ranks should be included. Each includes the nematodes as a single trophic group (Group 4). ITI G2b splits the nematodes equally into each of the four trophic groups. It can be seen that the latter procedure can have a dramatic effect on the index value. At the 0 m station the value rises from 48 to 84, a level that would be considered typical of normal undisturbed sediments. At the reference stations it rises from 15 to 55. This highlights another difficulty in interpreting this type of data set. Both reference stations are dominated by high populations of macrofaunal nematodes. This is atypical for normal sea loch sediments. Moreover, nearly 60 taxa and 900 individuals were recorded from the 500 m station but only 14 taxa totalling 200 individuals from the 800 m station. This disparity can be attributed to the problems associated with grab sampling in coarse substrates. The normal substrates in the vicinity of the farm are very coarse and poorly sorted, resulting in varying admixtures of clay, silt, gravel, shell and stone. Obtaining samples at all is difficult and successful samples are low in volume and obtained only from the slightly softer areas after repeated deployments of the grab. The 500 m reference station is probably more representative of the typical undisturbed community in the area, where brittle stars (*Amphiura, Ophiura*) and small bivalves (*Corbula, Abra*) predominate.

Table 1. Benthic survey results from a fish cage survey at a farm on the Scottish west coast. Dominant species present, population statistics and sedimentary characteristics at each station on a transect originating at the cages and at two reference stations.

Distance from cages	**0 m E**		**25 m E**		**50 m E**		**500 m W (ref.)**		**800 m W (ref.)**	
Station number	**T1**		**T2**		**T3**		**T6**		**T7**	
Dominant taxa	No.	Rank	No.	Rank	No.	Rank	No.	Rank	No.	Rank
Mytilus	108	1	5	10	-	-	-	-	-	-
Nematoda	23	2	85	2	26	5	602	1	158	1
Malacoceros	9	3	-	-	-	-	-	-	-	-
Harmothoe	2	4	-	-	-	-	-	-	-	-
Poecilochaetus	1	5	-	-	-	-	-	-	-	-
Skenea	1	5	-	-	-	-	-	-	-	-
Lacuna	1	5	-	-	-	-	-	-	-	-
Capitella	-	-	521	1	-	-	-	-	-	-
Cirrifomia	-	-	19	3	-	-	-	-	15	2
Thyasira	-	-	17	4	294	2	-	-	-	-
Chaetozone	-	-	16	5	95	3	-	-	-	-
Gyptis	-	-	15	6	-	-	-	-	-	-
Anaitides	-	-	14	7	-	-	-	-	-	-
Mediomastus	-	-	11	8	298	1	-	-	-	-
Scalibregma	-	-	10	9	16	8	5	9	-	-
Ophiodromus	-	-	5	10	12	9	-	-	1	9
Nemertea	-	-	-	-	37	4	-	-	1	9
Notomastus	-	-	-	-	26	5	-	-	-	-
Pholoe	-	-	-	-	25	7	5	9	-	-
Caulleriella	-	-	-	-	12	9	5	9	-	-
Amphiura	-	-	-	-	-	-	*127*	*2*	*3*	*5*
Tharyx	-	-	-	-	-	-	12	3	-	-
Nephtys	-	-	-	-	-	-	8	4	7	3
Nucula	-	-	-	-	-	-	7	5	-	-
Glycera	-	-	-	-	-	-	7	5	-	-
Prionospio	-	-	-	-	-	-	7	5	2	6
Corbula	-	-	-	-	-	-	7	5	-	-
Aricidea	-	-	-	-	-	-	5	9	-	-
Ophiura	-	-	-	-	-	-	5	9	-	-
Abludomelita	-	-	-	-	-	-	-	-	5	4
Abra	-	-	-	-	-	-	-	-	2	6
Spiophanes	-	-	-	-	-	-	-	-	2	6
Eulalia	-	-	-	-	-	-	-	-	1	9
Lumbrineris	-	-	-	-	-	-	-	-	1	9
Orbinia	-	-	-	-	-	-	-	-	1	9
Sediment type	Dark grey soft mud		Grey soft mud		Grey soft mud		Grey/brown sandy mud		Grey/brown mud	
Feed/faeces	✓		-		-		-		-	
Gas bubbling	-		-		-		-		-	
H_2S smell	✓		✓		-		-		-	
Redox (40 mm depth)	-195		-52		+108		+73		+99	
Oxygen (mg/l)	6		6		4.5		6.5		-	
Depth (m)	39		39		36		44		34	
Total taxa (S)	7		28		27		59		14	
Total abundance (A)	145		732		882		877		200	
H'	0.5		1.87		1.84		1.35		1.03	
J	0.83		0.63		0.24		0.38		0.42	
ITI (MH)	48.3		19.8		48.2		19.3		15.1	
ITI (SEASa)	48.2		10.9		50.1		15.1		15.5	
ITI (SEASb)	84.3		7.0		51.4		55.7		53.5	
Distance from cages	**0 m N**		**25 m N**		**50 m N**		**500 m W (ref.)**		**800 m W (ref.)**	
Station number	**T1**		**T2**		**T3**		**T6**		**T7**	

A comparison of the ITI values at these reference stations sampled over a number of years suggests that the maximum ITI value for a normal sediment in the area, not dominated by nematodes, would be between 40 and 60, rather than a value of between 60 and 100 that is suggested as 'normal' for undisturbed sediments in the SEPA guidelines.

These difficulties suggest that placing reliance on ITI values to differentiate between these samples would be very misleading. In this case when the ITI is calculated with the nematodes as a single group the cage and 50 m stations would be listed as ‘changed’ and the 25 m station as ‘degraded’ in the SEPA classification. When calculated with the nematodes split the first station would be classified as ‘normal’, the 25 m as degraded and the 50 m as ‘changed. Both reference stations would be classified as ‘degraded ‘ using the first type of calculation and as ‘changed' by the second way. It should be note that none of the three ways of calculating the ITI used here contravene the rather loosely written protocols established for its calculation. Comparative assessment of the other diversity indices and the distribution of faunal dominants would suggest that both the 0 and 25 m stations were ‘degraded’ and the 50 m station and reference stations were ‘normal’. This re-emphasises the need to take account of as wide a range of data interpretations as possible when interpreting faunal response to environmental change.

Multivariate approaches

Emphasis has so far been put on the use of univariate data and indices to assess benthic faunal response to environmental disturbance. In recent years, as statistical techniques have improved increasing emphasis has been put on the use of multivariate statistics to assess such response and to associate it quantitatively with varying levels and types of disturbance. Both CGMSD and SEPA guidelines recommend the use of hierarchical and ordination analyses in assessments of benthic faunal data sets. However such analyses are most effective in the analysis of extensive data sets based on many samples. In such cases the techniques provided in, e.g. the PRIMER (Clark 1993, Clarke & Ainsworth 1993) and CANOCO (Jongman *et al*. 1987) statistical packages allow direct linkages to be made between biological variance and changes in specific environmental factors, including the ranking of such factors in order of importance. Another recent innovation has been the development of a more objective quantitative index to measure the condition of benthic assemblages with defined thresholds for levels of environmental disturbance (Smith *et al*. 1997). This Benthic Response Index (BRI) is calculated using a two-step process. Initially ordination analysis is used to quantify a pollution gradient within an appropriate database of samples taken from as wide a range of conditions as possible. The pollution tolerance of each species is then determined based on its abundance distribution along that gradient. The index is calculated as the abundance weighted average pollution tolerance of species in a sample. Thresholds are established for the reference condition as well as for four levels of biological response, identified as the index values at which key community attributes are lost. The BRI was developed and validated with data from the southern California coastal shelf, when it was recognised that existing indices (including the Word index from which the ITI is derived) were too subjective in their assignment of pollution tolerance scores. Moreover the direct use of diagnostics from multivariate analyses to categorise response states was considered too complex for use in the coastal management reports. The BRI is considered to work well in Californian coastal impact assessments and should be relatively easy to adapt to conditions in UK coastal areas where adequate data sets are available to calibrate gradients and validate the calculations.

Progress in model building

A number of models have been developed to provide predictions of the environmental impact of coastal disturbance, e.g. Cromey *et al.* 1998, Henderson *et al*. 2001. The BenOss model and it's later development DEPOMOD predict the likely distribution of suspended particles from a discharge source based on a knowledge of particle form and density and the local hydrographic regime (Cromey *et al*. 2000). This has been recommended for use in coastal impact assessments by both the CGMSD and SEPA. A further development of this model will add a benthic module to predict the probable extent of non-compliance with EQSs based on ITI values.

In view of the problems discussed above it is recommended very strongly that the module should not be based on the ITI values. There are many much more specific and direct indices of faunal change which could be used as faunal data inputs to the model. It is suggested that, where available data is limited use should be made of the most direct statistics possible, i.e. T, the total number of species (taxa) in a sample, A, the total faunal abundance of a sample, or, as a composite diversity index, their ratio A/S. These simple univariate statistics are known to vary in predictable ways in response to carbon enrichment (Pearson & Rosenberg, 1978; Pearson, 1987) and are direct quantitative values that can not be misinterpreted prior to their use in a modelling exercise. Where extensive data sets, sufficient to quantify and validate the BRI, are available then the use of that index may be more effective.

Best practice and continuous review and development

It is evident from the above account that the scientific basis underlying the development and use of summary indices of benthic response to environmental impacts is undergoing rapid development. Both environmental practitioners and managers must be aware that best practice in this field is continually being updated and that guidelines must be regularly updated as new field and analytical technologies become available.

Conclusions and recommendations

It has been pointed out that there are a number of worrying deficiencies in the way the ITI is calculated and used. Further validation of it's application in coarse sediment areas should be carried out and it should not be used in isolation when setting or applying biological standards for the control of pollution. There are a number of other, simpler indices, e.g. A/S (the mean number of individuals per taxon in a sample) that were shown to be as informative as the ITI during the validation process of that index (WRc plc, 1992). It is suggested that further evaluation of the utility of the ITI should be made. Meanwhile more emphasis should be given to judging impacts on the basis of changes in a suite of factors rather than on the comparative values of a single index.

Setting a standard on the basis of a single index that is intended to summarise the condition of the biological system in its entirety is poor practice. In this case, when that index is clearly unrepresentative in some environmental conditions, is flawed in the way the data is derived and can be clearly altered by re-interpreting some of the key data inputs, then it should not be used to set a biological standard.

It is recommended that the BRI recently developed for assessments on the southern California shelf, should be adapted to UK use where appropriate validation data is available.

References

CROMEY, C.J., BLACK, K.D., EDWARDS, A. AND JACK, I.A., 1998. Modelling the deposition and organic effects of organic carbon from marine sewage discharges. Estuarine, Coastal and Shelf Science, *47*: 295-308.

CROMEY, C.J., NICKELL, T.D. AND BLACK, K.D., 2000. DEPOMOD (v1.5)software: a model for predicting the effects of solids deposition to the benthos from mariculture. CCMS Dunstaffnage Marine Laboratory, P.O. Box 3, Oban, Argyll UK, PA34 4AD.

CLARK, K.R., 1993. Non-parametric multivariate analyses of changes in community structure. Australian Journal of Ecology, *18*: 117-143.

CLARK, K.R. AND AINSWORTH, M., 1993. A method of linking multivariate community structure to environmental variables. Marine Ecology Progress Series, *92*: 205-219.

CSTT, 1994. MPMMG. Comprehensive studies for the purposes of Article 6 of DIR 91/271 EEC, the Urban Waste Water Treatment Directive. Forth River Purification Board, Edinburgh. 42 pp + figs.

CSTT, 1997. MPMMG. Comprehensive studies for the purposes of Articles 6 and 8.5 of DIR 91/271 EEC, the Urban Waste Water Treatment Directive. Department of Environment for Northern Ireland, The Environment Agency, the Scottish Environment Protection Agency and the Water Services Association. 54 pp + figs.

HENDERSON, A.R., GAMITO, S., KARAKASSIS, I., PEDERSEN, P.B. AND SMAAL, A.C., 2001. The use of hydrodynamic and benthic models for managing environmental impacts. Journal of Applied Ichthyology, *17*: 163-172.

HENDERSON, A.R. AND ROSS, D.J. 1995. Use of macrobenthic infaunal communities in the monitoring and control of the impact of marine cage fish farming. Aquaculture Research, *26*: 659-678.

MEARNS, A.J. & WORD, J.Q., 1982. Forecasting effects of sewage solids on marine benthic communities. In Mayer, G. F. (ed.) Ecological Stress and the New York Bight: science and management. Columbia, S. Carolina: Estuarine Research Federation, pp 495-512.

PEARSON, T.H., 1987. Benthic ecology in an accumulating sludge disposal site. In, *Biological Processes and Waste in the Ocean*. Oceanic Processes in Marine Pollution, Vol. 1. Edited by J.M. Capuzzo and D.A. Kester, Robert E. Krieger Publishing Co., Malaba, pp. 195-200.

PEARSON, T.H. AND ROSENBERG, R., 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. *Oceanography and Marine Biology Annual Review 16*: 229-311.

SMITH, R.W., BERGEN, M., WEISBERG, S.B., CADIEN, D., DALKEY, A., MONTANGE, D., STULL, J.K. AND VELARDE, R.G., 1997. Benthic response index for assessing infaunal communities on the mainland shelf of Southern California. http://www.sccwrp.org/pubs/annrpt/97/ar.15.htm

WORD, J.Q., 1979. The Infaunal Trophic, Index. In: Coastal Water Research Project Annual Report 1978. Southern California Coastal Water Research Project, El Secundo, California, pp. 18-39.

WRC PLC, 1992. Development of a biotic index for the assessment of the pollution status of marine benthic communities. Final Report NR 3102/1. 78 pp.

A comparison of the values of the major macrofaunal population statistics for stations at varying distances from a sewage outfall discharging into the entrance of a sea small loch on the Scottish west coast. Stations 1-6 lie between 100-200 m from the outfall; stations 13, 16 and 17 lie outside the loch between 2-3 km from the outfall; stations 23-25 lie in the inner loch between 1.5-3 km from the outfall. Data from the autumn and winter surveys are presented.

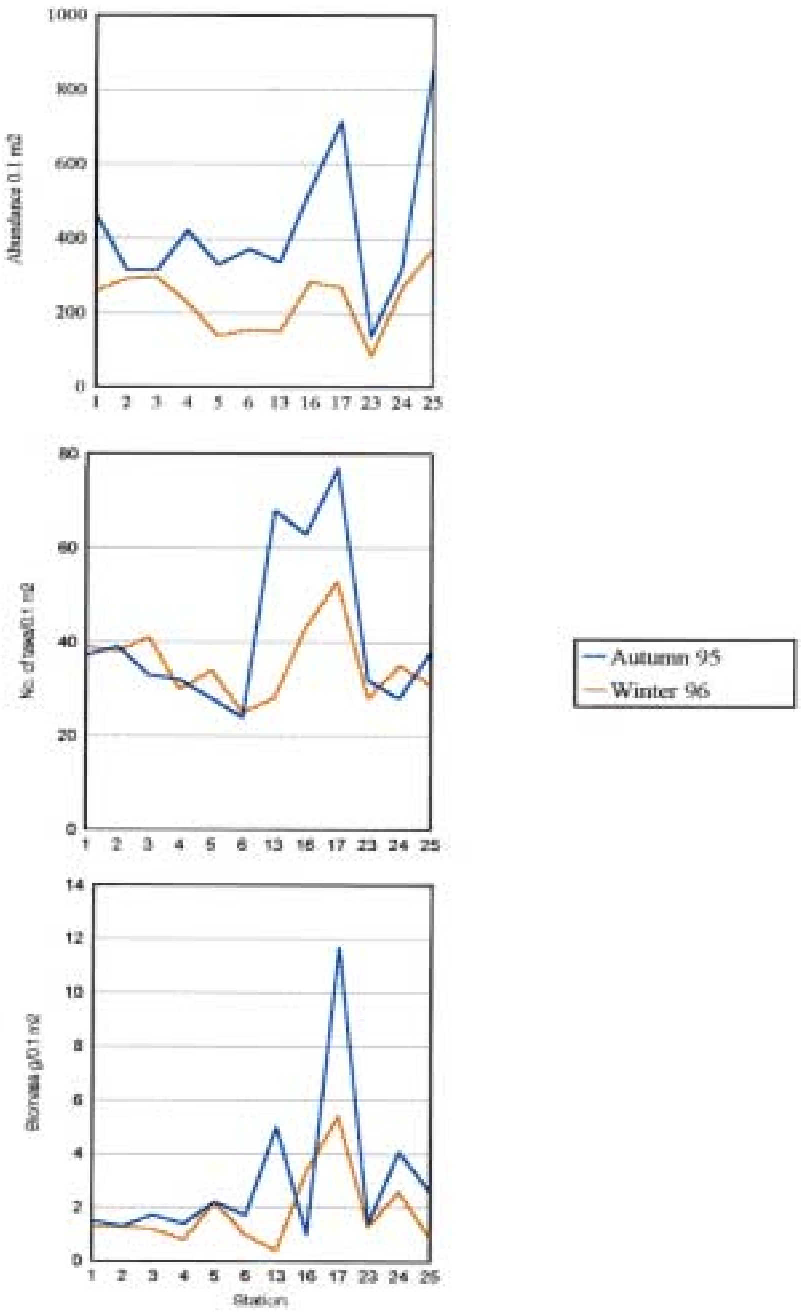

Figure 1. Abundance, number of species/taxa and biomass of benthic fauna in samples collected during the Autumn 1995 and Winter 1996 surveys

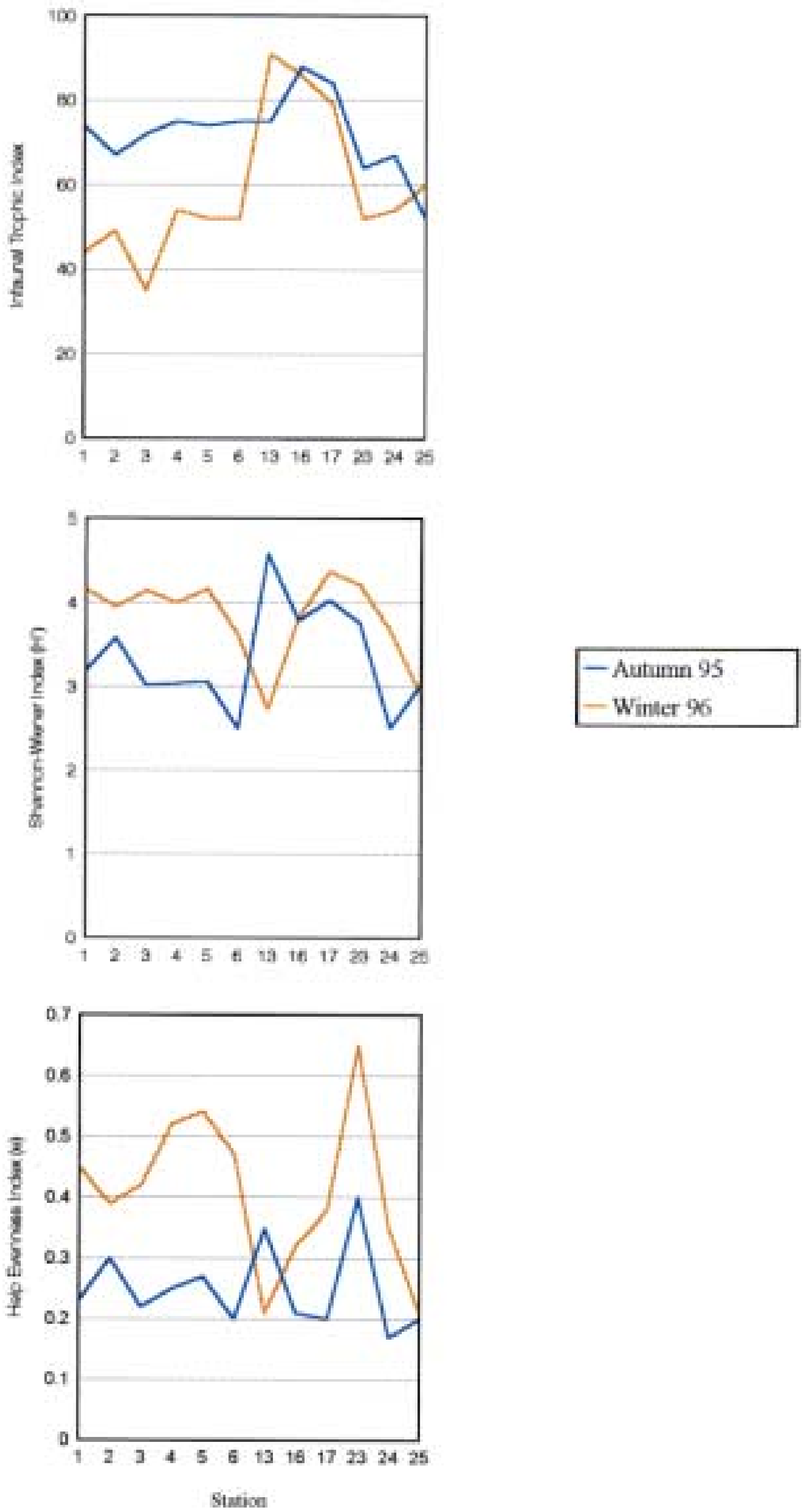

Figure 2. Indices derived from the benthic population data

Application and Applicability of Ecological Objectives and Standards for Dredging and Aggregate Extraction

M. Elliott
Institute of Estuarine & Coastal Studies
University of Hull, HU6 7RX, www.hull.ac.uk/iecs
E-mail: Mike.Elliott@hull.ac.uk

Derivation of EcoQO's and history of approach

EcoQO have been defined as an overall expression of the structure and function of aquatic systems and thus they are an expansion of Environmental Quality Objectives. They were developed from the EQO/EQS approach whereby environmental statements (i.e. objectives) were derived as the desired outcome for an area, e.g. that an area should support the passage of migratory fishes. These could be phrased in the form of Null Hypotheses whereby field surveys could be carried out to test whether they were met (Costa & Elliott 1991). The EQO were accompanied by standards (EQS) as numerical values against which sampling could then be used to see if they were met, i.e. as a level of compliance. Within the UK, the EQO/EQS approach accompanied the consenting/authorisation/licensing procedure, i.e. the discharge permit was set in order to ensure that EQO were met and that EQS were used to inform the monitoring. This approach relied on there being a good relationship between the EQS being met and the EQO being achieved. For example, if organic enrichment is prevented such that dissolved oxygen does not reach low levels then fish will be able to live and migrate through the waters.

The EQO were derived to relate to biological features, such as the ability of an area to support predators, or human uses of the area, such as the ability to maintain a bathing beach. EQS classically were environmental parameters, such as concentrations of trace metal in a medium (biotic, e.g. fish flesh, or environmental, e.g. seawater concentrations). Field surveys are then carried out to determine whether a standard is met with the assumption that management action is taken if standards are not met. This is regarded here as monitoring *sensu strictu* and thus is distinguished from surveillance, the latter is whereby field surveys are carried out with a *post hoc* detection of trends followed by the explanation of those trends.

The relationship between Ecological and Environmental Quality Objectives and Standards (EcoQO/EcoQS/EQO/EQS) was discussed in the earlier paper by Elliott (1996) and by Lanters *et al.* (1999). The change of term from EQO to EcoQO is a more accurate representation of the term whereby biological and ecological health-related objectives are required. Thus there should be a separation of these from true Environmental Quality Objectives, i.e. those relating to the human use of the system such as the requirement for an area to support bathing. The extension of Environmental Quality Standards to Ecological Quality Standards requires a large degree of development in that EQS have usually been derived for chemicals, including dissolved oxygen, ammonia, trace metals, whereas EcoQS should be for ecological or biological health variables. The only previous case of biological EQS have been for microbiology, e.g. levels of faecal coliforms permitted under the EU

Bathing Beach Directive although those for benthic macrofaunal community have been proposed (MAFF 1993; SOAEFD 1996; also Elliott 1996).

Previous use of EcoQO/S

There are few easily accessible examples of EQO and EQS being incorporated into legislation and statutory frameworks (e.g. see Ducrotoy and Elliott 1997) although the UK Water Act 1989 has provision for statutory objectives and standards. To date, the only ones with legislative support are those related to European Directives although the UK environment protection agencies have locally adopted such standards (ref. to Environment Agency's State of the Coast Report).

The EA in England and Wales uses non-statutory estuary quality objectives as a basis for water quality management and licensing. In essence, the OSPAR agreement (at Sintra, 1998) by signatories, to achieve concentrations near background levels for natural substances and at zero for synthetic substances, is also a use of environmental quality standards, whereby discharge controls and monitoring are directed towards achieving the standards.

The Dutch RIKZ have adopted operational objectives and are now developing criteria for judging eutrophication, its causes and consequences (report not yet publicly available) and Scott *et al.* (1999) have detailed the features and consequences of nutrient enrichment in estuaries as an example of the use of such objectives in one habitat.

Differentiation between objectives and standards

In essence, this is the separation between statements (objectives) versus numerical values (standards) thus following the EQO/EQS approach whereby objectives require accompanying standards in order to provide an end-point for effective monitoring (see below regarding the Reference state). Consequently, it is difficult to produce monitoring programmes or to see if an environmental quality is met unless standards are produced and accompanied by rigorously defined monitoring protocols. For example, with regard to aggregate extraction:

- an objective may be to ensure that the sediments supports an adequate biomass as fish food,
- the accompanying standard or reference will be that <x% of an area should have biomass of y g m^{-2} as indicated by its sediment nature, salinity and depth.
- the monitoring is designed with sufficient rigour and frequency to detect such a change,
- and management actions are then taken if the standard is breached.

Format for derivation of ecological objectives

As an example, the signatories to OSPAR have adopted a Common Procedure for the identification of the eutrophication status of the maritime areas. This requires the implementation of integrated target orientated and source-orientated actions for problem areas, including the determination of Ecological Quality Objectives which OSPAR regard as 'the desired level of ecological quality relative to the reference level'. Hence the requirement by the UK, following the Common Procedure, is to develop EcoQO and their accompanying EcoQS (as interpreted here in relation to the reference levels) for the components at risk, as early warnings of change or the end point of change. While any system may show the low level effects of change, it is considered that many systems have an inherent capacity for absorbing that change ('environmental homeostasis'). Thus it is perhaps of greater significance and often more meaningful to take as the end point the aspects of particular socio-economic significance or those significant in human perception, e.g. aesthetic aspects. The derivation of these indicators will require to be divided into structural and functional components, i.e. respectively attributes at a single census point and rate processes. EcoQS, as the monitoring end-points, will thus have to be derived around these structural or functional end points.

Identification of dredging and dredged-material deposition on marine systems and current state of knowledge

The discussion will include:

- conceptual model of change (see Figures 3 & 4);
- bottom-up processes and top down responses;
- confidence in ascribing effects to sediment modification drivers; and
- ecological web interactions.

There may be insufficient case studies, especially where all elements of the process have been studied, to fully quantify all the links in the sequence of change. Furthermore, modelling approaches, whereby predictions are attempted to give change resulting from cause, are not yet sufficient for whole ecosystems. There are, however, some good attempts to move in this direction (e.g. the BOEDE model for the Ems-Dollard) and the CSTT approach used in the UK to denote sensitivity of receiving areas under the Urban Waste-Water Directive (Elliott *et al.* 1999). There is an increasing movement towards the holistic assessment of systems with regard to their health, for example the use of coastal and estuarine classification schemes as indicators or change and as management tools (ADRIS/SEPA unpublished).

The Scheveningen (Netherlands) workshop organised by OSPAR in September 1999 proposed a set of 10 issues for developing EcoQOs for the North Sea (Table 2). In order to take these forward, the approach could then be expanded to include the different signs/symptoms relating to dredging and material relocation which in turn may then developed to form the individual EcoQO and their related EcoQS.

Table 2 Proposed Issues for Developing EcoQO

No.	Proposed Issue	No.	Proposed Issue
1	Reference points for commercial fish species	6	Plankton communities
2	Threatened or declining species	7	Benthic communities
3	Sea mammals	8	Habitats
4	Sea birds	9	Nutrient budget and production
5	Fish communities	10	Oxygen consumption

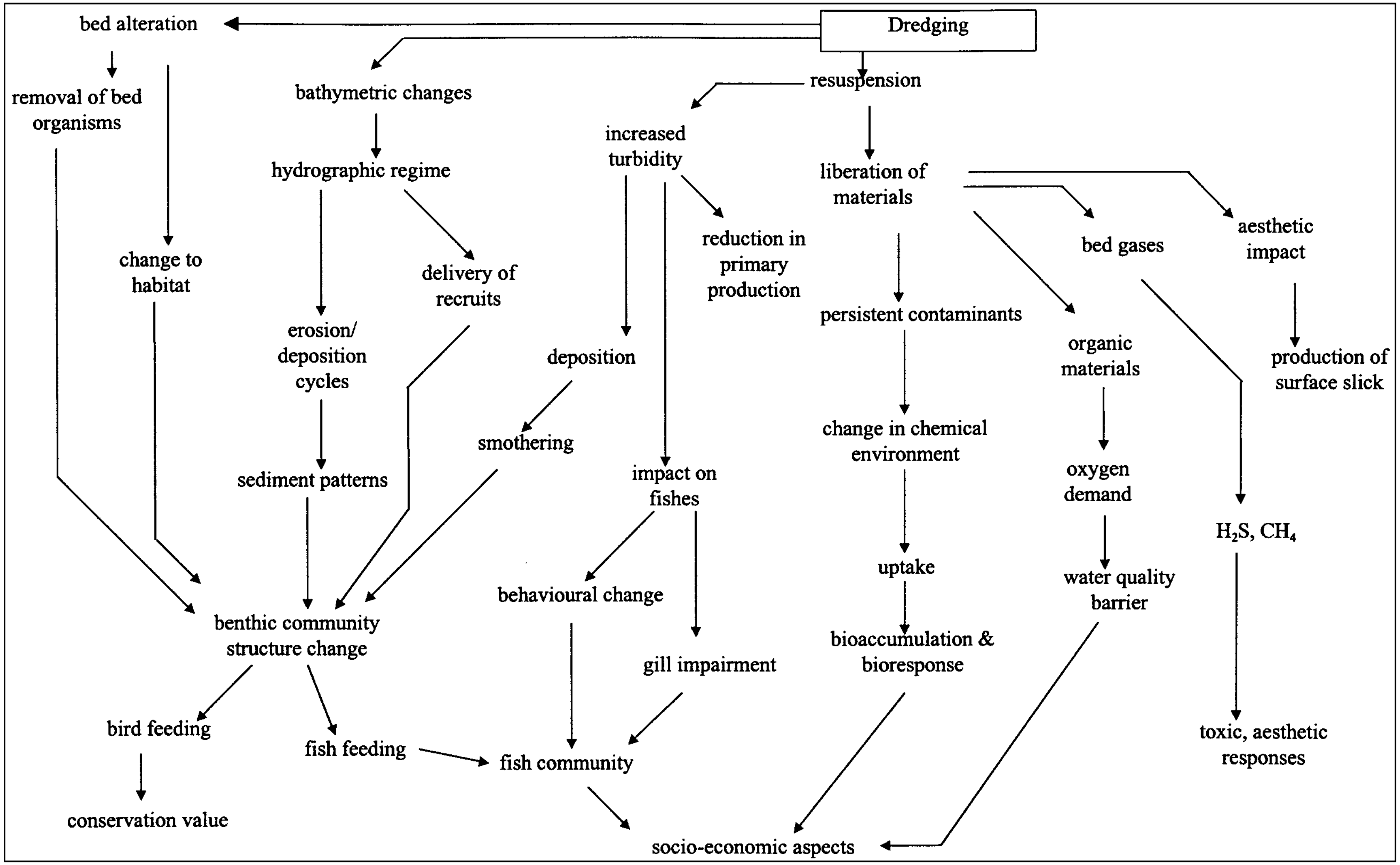

Figure 3 Conceptual Model of Change - Dredging Impacts

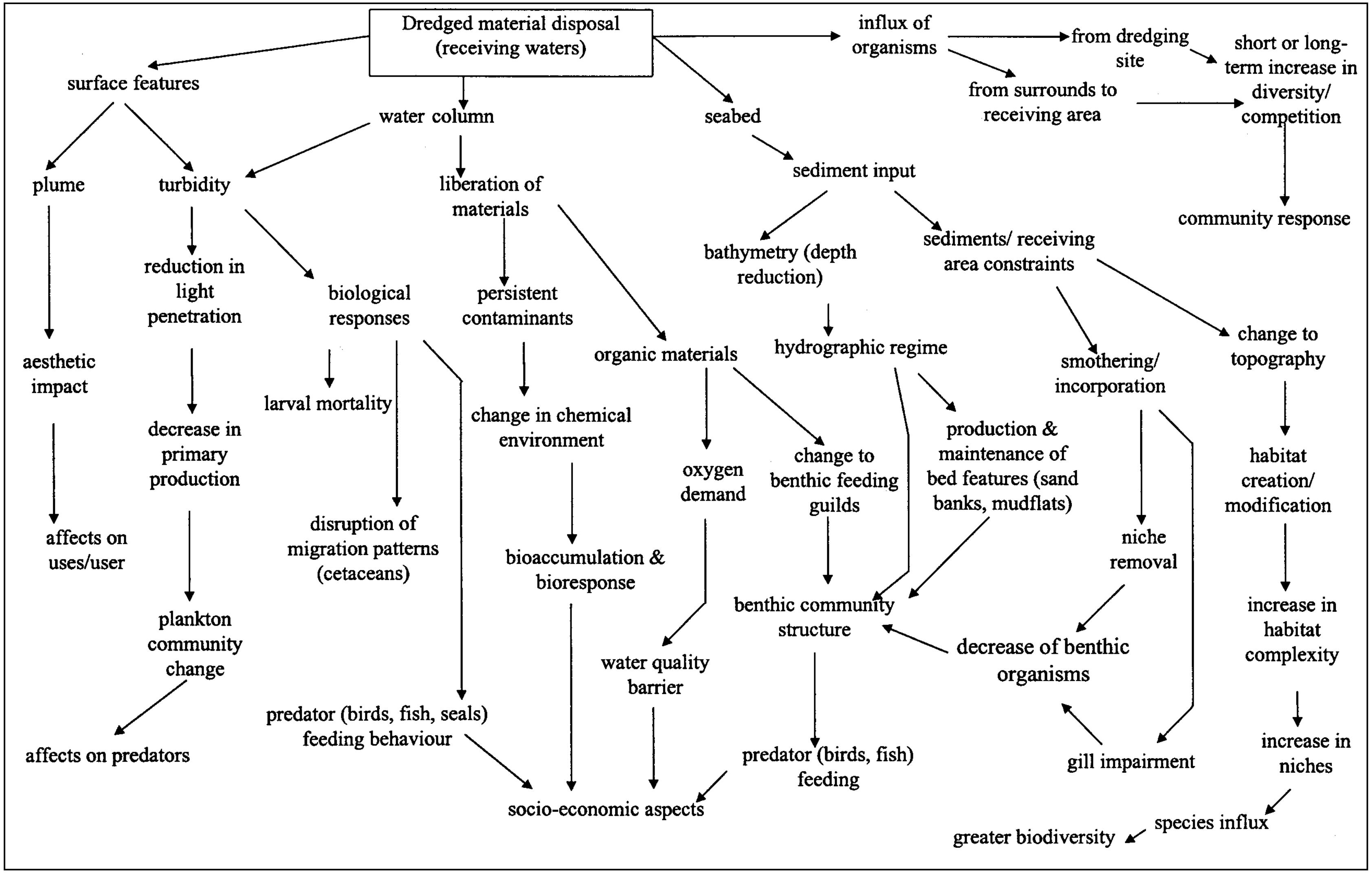

Figure 4 Conceptual Model of Change - Dredged Material Disposal

References

COSTA, M.J. & ELLIOTT, M., 1991. Fish usage and feeding in two industrialised estuaries - the Tagus, Portugal, and the Forth, Scotland. In: Elliott, M and J-P Ducrotoy (Eds.), *Coasts and Estuaries: spatial and temporal comparisons*. Olsen & Olsen, Fredensborg, Denmark, 289-297.

DUCROTOY, J.-P. & ELLIOTT, M., 1997. Interrelations between science and policy-making: the North Sea example. Marine Pollution Bulletin *34*(9): 686-701.

ELLIOTT, M., 1996. The derivation and value of ecological quality standards and objectives. Marine Pollution Bulletin *32*(11): 762-763.

ELLIOTT, M., FERNANDES, T.F. & DE JONGE, V.N., 1999. The impact of recent European Directives on estuarine and coastal science and management. Aquatic Ecology *33*: 311-321.

ELLIOTT, M., NEDWELL, S., JONES, N.V., READ, S., CUTTS N.D. & HEMINGWAY, K.L., 1998. Intertidal sand and mudflats and subtidal mobile sandbanks (Volume II). An overview of dynamic and sensitivity characteristics for conservation management of marine SACs. Scottish Association for Marine Science, Oban, for the UK Marine SAC project, pp151.

LANTERS, R.L.P., SKJOLDAL, H.R. & NOJI, T.T., 1999. Ecological Quality Objectives for the North Sea. RIKZ/Institute of Marine Research, Report 99.015, The Hague.

MAFF, 1993. Analysis and interpretation of benthic community data at sewage sludge disposal sites. Aquatic Env. Mon. Rept. 37, Lowestoft, pp80. (2nd report of the Biology Task-team, GCSDM/MPMMG/DOE, includes two sections with ME as author plus other parts drafted by whole team).

SOAFD, 1996. Monitoring and assessment of the marine benthos at UK dredged material disposal sites. Scot. Fish. Inf. Pamph. No. 21, Aberdeen, ISSN 0309 9105 (3rd report of the Biology Task-team, GCSDM/MPMMG/DOE).

Annex: ELLIOTT, M., 1996. The derivation and value of ecological quality standards and objectives. Marine Pollution Bulletin. *32*(11): 762-763

All Environmental Impact Assessments centre on the potential reduction in the health of the system as the result of human activities. As the main questions being asked relates to the health of the whole system, then it is axiomatic that the system should be studied at the higher levels of biological organisation - the ecosystem, community and population levels. Furthermore, there is the continuing demand by environmental managers that biologists assessing quality and predicting changes as the result of human activities make such assessments and predictions fully quantified. Despite this, the use of macrobiological information is made difficult by the inherent complexity of the system in coastal and estuarine areas.

In several countries, and despite moves towards a Precautionary Approach, the assessment and control of pollution centres around an approach based on the ability of the receiving waters to which pollutants are discharged to degrade, disperse and assimilate those pollutants. For example, pollution control authorities in the UK have used to good effect the derivation and then adoption of statements (Environmental Quality Objectives, EQO) based on the uses to which a water body is put. For the purposes of biological assessment, these objectives with regard to the health of the system can be framed as Ecological Quality Objectives (EcoQO) and in turn null hypotheses (Table A) and thus they provide the basis against which monitoring can be carried out. In essence, testing by field survey and/or field and laboratory experiments can then assess whether such hypotheses should be rejected thus indicating degradation of the marine environment.

The EQO approach requires to be accompanied by the derivation and adoption of EQS, Environmental Quality Standards. These are numerical values which if achieved should ensure that EQO are met although a valid EQS is dependent on the relationship between it and the EQO being well-understood. Following the derivation of an EQS, and in contrast to surveillance, monitoring is then carried out against the background of this *a priori* derived standard. Many chemical-based and water column EQS have been derived and adopted, *e.g.* the EQS for dissolved oxygen in estuaries of 5 mg/l to allow the passage of migratory salmonid fishes, the limit of 0.3 mg/kg Hg in fish flesh aimed at protecting consumers of the fish, and the faecal coliform levels of 2000 per 100 ml of water to protect bathers.

Thus EQS have usually been chemically or at best microbiologically based and as yet there are few EQS based on macro-biological parameters. This is an important omission as the questions to be answered in impact assessment relate to the health of the whole system and then any standards should relate to that system. Similarly, although many pollution problems and assessments centre on changes to the bed sediments which may be the ultimate reservoir of the contaminants, the derivation of sediment biological, *i.e.* benthic, and the accompanying sediment chemical standards are only now being considered. The lack of such standards is of concern especially as there are increasing cases whereby water quality is sufficiently high but seabed quality is degraded.

Table A. Ecological Quality Objectives (EcoQO) and Null Hypotheses

1	The water quality of an estuarine area always allows the passage of migratory fish.
2	The structure of the resident marine/estuarine fish assemblage and populations are consistent with the hydrophysical regime.
3	The benthic populations and sediments are of a quality sufficient to support the fish and (when necessary) bird populations.
4	The structure of the intertidal/subtidal benthic (soft-substratum) community and populations are consistent with the hydrophysical regime.
5	The diversity, abundance, biomass and population structures of the intertidal and subtidal rock community are as expected given the physical features of the area.
6	The planktonic communities have not been/will not be unacceptably affected by man's activities.
7	The levels of potentially pathogenic micro-organisms in sea-water and biota are insignificant, do not affect their users or exceed the statutory or agreed limits.
8	The biological functioning of an area has not been/will not be changed unacceptably by waste disposal or other anthropogenic changes.
9	The levels of persistent toxic and tainting substances in the biota are insignificant and do not affect its biology, it being predated or the health of its predators including man.
10	The concentrations and body burdens of toxic and tainting substances either agreed or defined in conventions and legislation are not exceeded in the relevant biological component.

Until recently, there have been few attempts to derive standards summarising the quality or indeed any degradation at the community level. In particular, in carrying out environmental impact assessments, biologists are being required to predict quantitatively the degree of change in the benthos as a result of human activities. For example, environmental managers require to know in advance the amount of change to the benthos following construction and operation of effluent discharge pipelines or the permissible change in areas exposed to dredged-material disposal. Because of this, biologists are being asked to derive ecological quality standards (EcoQS) against which monitoring should then be carried out.

The derivation of EcoQS requires that the normal situation is defined and that its limits of variability are quantified. Following this, the assumption is that a given signal, as a change from the normal situation and allowing for spatial and temporal changes, will be observed and measured. In general terms, if such a signal deviating from the background noise is predictable then a numerical quality standard can be derived. However, the practice is very much different from the theory - we biologists have to question whether we understand the system sufficiently well to indicate rigorously the magnitude of any expected change under a given anthropogenic perturbation. Previously, any predictive capability has been largely subjective although objective assessments have been made of the possible change in community composition. We have good conceptual models of change but only recently are becoming sufficiently confident to derive fully quantitative predictions for communities.

In essence, we have a good conceptual knowledge but are less sure quantitatively such that while we are happy to say in which direction a community will change, we are more reluctant to give the amount of change and thus derive the numerical standards. We are now required to start being more confident in giving the degree of (*i.e.* numerical) change to the number of species expected, the structure, functioning and components of the community, the behavioural and physiological response, and the expected population structure and dynamics following any human-induced changes.

As an indication of the way ahead, the (now completed) Biology Task-team of the UK Group Co-ordinating Sea Disposal Monitoring was given the task of deriving such community EcoQS and the ongoing UK Dredged Material Disposal Monitoring Task-team is also attempting to derive standards. The former recently proposed tentative benthic EcoQS for testing of the changes in benthos at sewage sludge disposal sites; these are centred on changes in primary and derived community parameters for comparable reference and treatment sites. The initial changes against baseline values for the primary variables are: abundance (+/- 50%), taxa (+/- 20%), and biomass (+/- 20%), and for the derived variables H' (Shannon-Wiener information statistic, +/- 20%), A/T (abundance ratio, +/- 50%) and B/A (biomass ratio, +/- 50%). These values were derived empirically from studies of severely impacted areas; the concepts are currently being developed further and the statistics of sampling and compliance being considered. Thus these EcoQS have been developed for a well-defined response, that of organic enrichment on benthic conditions, but a poorer background knowledge may delay or prevent them being derived for other effects.

The message here is that just because it is difficult task or gives less than precise results, this should not deter marine impact assessment ecologists from attempting to derive standards. We should ensure that EcoQO and EcoQS are derived and incorporated into quality assessments. We have a choice - *either* to develop EcoQS as good as is presently possible and to allow their use by managers but with the provisos that the science is far from perfect and that caution should be exercised, i.e. they are a guide to further studies. *Or* to let the ecological input be used less effectively and to let marine quality management continue to rely heavily on a chemical contaminant basis.

Practical considerations relating to the derivation of Ecological Quality Objectives for marine aggregate extraction activities

H.L.Rees
Centre for Environment Fisheries and Aquaculture Science
Burnham-on-Crouch, Essex, CM0 8QD
E-mail: H.L.Rees@cefas.co.uk

This paper highlights the potential advantages and pitfalls arising from the application of an EcoQO approach, by specific reference to sampling practices at marine aggregate extraction sites. These are presented as a series of discussion points to facilitate progress in subsequent Workshop activity.

The Ideal EcoQO

An EcoQO will, ideally, be:

1. quantifiable and statistically robust, with minimal associated sampling effort;
2. simple in construction, conveying a readily-understood message;
3. underpinned by a foolproof scientific rationale for its behaviour, *i.e.* the cause/effect relationship will be absolute and unambiguous;
4. responsive in a predictable and, preferably, linear way to variation in impact;
5. insensitive to natural change;
6. relevant, if not directly, then by quantifiable association with matters of real concern such as population or community health, habitat or wider ecosystem integrity, and the performance of commercial fisheries; and
7. defensible in a court of law, and therefore suitable for use as a statutory enforcement tool in regulating the environmental consequences of extraction activity.

The Reality

How far short are we from the ideal, and what needs to be done to advance the prospect for application of an EcoQO approach in routine environmental management? These questions are dealt with by reference to the above list of seven desirable attributes, as they relate to conditions at the seabed (*i.e.* sediments and the associated benthic biota) since this represents the most logical target, when considering the effects of marine aggregate extraction.

1. Quantifiable and statistically robust, with minimal associated sampling effort

In order to satisfy immediate requirements, the measure will need to utilise data arising from current sampling practices. In recent years, the sampling of bottom sediments in environmental surveys has employed grabs, cores, dredges and trawls. Non-destructive sampling tools include the use of underwater photography (by diver, remotely-deployed frame support or Remotely Operated Vehicle) and acoustic ground discrimination systems.

The deployment of all these tools is intrinsically straightforward, but their performance, and hence reliability, can vary with substratum type, sea state and turbidity. The scope for accurate and quantitative characterisation of seabed substrata is, in general, greater than for the biota, which may have a bearing on the target for investigation, since an EcoQO need not involve biological measurement, provided that there is some dependency between changes in the measure and in biological systems.

Presently, the Hamon grab is the most reliable device for retrieving quantitative samples of the seabed fauna and associated sediments in environmental surveys. It is reasonably versatile, allowing acceptable sampling from a range of unconsolidated gravely substrata in (up to) moderate sea states. In common with many other sampling devices, its efficiency decreases in marginal weather conditions and in the presence of very coarse or consolidated substrata. This would suggest that the use of such a device for generating data of use in an EcoQO context should be confined to locations and circumstances where the performance is dependable.

The maintenance of an acceptable level of consistency during field sampling, and in subsequent laboratory analyses, may be achieved through the application of standard AQC procedures. One feature of the benthos typically associated with gravely deposits is the increased frequency of colonial organisms, which are not amenable to counting in the same manner as for solitary species. To meet the requirements of an index utilising species densities, the information content may therefore have to be reduced through the exclusion of such species. Biomass data may also be usefully employed in an EcoQO context, although present experience suggests that between-laboratory consistency in determinations can be unacceptably low.

Small-scale patchiness in the distribution of sediment types from areas of marine aggregate is commonly encountered; in such circumstances, relatively large numbers of samples may be required to meet the statistical requirements of an EcoQO approach. This may, to some extent, be ameliorated by appropriate site selection. Thus, on grounds of cost-effectiveness, an EcoQO may be derived in order to address the well being of a *component* of the target environment, rather than its entirety. Consideration should also be given to the availability and practicality of alternative quantitative sampling devices, *e.g.* adaptations of larger hydraulically operated grabs used by the industry in resource surveys.

The Hamon grab is an effective sampling tool for the infauna and smaller, sessile or less mobile epifauna. Sampling of the larger, rarer or more mobile epifauna using small trawls can provide important additional information concerning the well being of the benthic habitat. However, the data are, at best, semi-quantitative in nature, and large numbers of samples may be needed to satisfy the statistical requirements of a ‘quality’ index. Small trawls may also operate with varying (but always relatively low) sampling efficiency and may, in areas of coarser substrata, be unsafe to deploy. This suggests that sampling in an EcoQO context should be confined to dependable locations.

Improvements in the resolution of devices for acoustic surveys of the seabed, as well as an increase in the variety of available methods, offer scope for future derivation of indices of seabed (and biotic) condition, and will merit Workshop attention.

Consideration should be given to the utility, and reliability, of qualitative biological data in the derivation of EcoQOs, and of measures additional to animal counts, such as age structure or growth rates of target species, as expressions of quality status.

Underwater photography has the potential to generate reliable data on the status of the superficial seabed fauna and sediments. Routine applications may be hindered by variability in visibility and sea state. Nevertheless, methodology for accurate and consistent recording of the contents of seabed images may merit further consideration in relation to EcoQO derivation.

For samples of sediments, standard laboratory procedures for particle sizing allow for accurate quantification. EcoQO derivations may therefore be useful, provided that field sampling is adequate, and that the information is interpretable in a wider ecosystem context. Consideration should also be given to the utility of vibro-core sampling in an environmental monitoring context, as a means to evaluate the vertical integrity of sediments in response to aggregate dredging and its aftermath.

2. Simple in construction, conveying a readily understood message

This presents a considerable challenge to field biologists, since univariate measures of community status may be more readily communicable (and more amenable to statistical analysis), but less sensitive than the more complex outcomes of multivariate analysis. Previous experience suggests that the use of *combinations* of summary measures, each conveying a different attribute of community structure or function, may be the best way to proceed. An evaluation of the utility of available measures will be an important area for consideration during the Workshop.

3. Underpinned by a foolproof scientific rationale for its behaviour, i.e. the cause/effect relationship will be absolute and unambiguous

Other presentations provide insight into the current state of knowledge regarding the nature of biological impacts, and the suitability of different measures. A conceptually simple and universally applicable measure of man-made impacts may be considered as the ‘holy grail’ of applied environmental science and, as yet, the scientific community remains some way short of achieving this. Nevertheless, we have seen that some advances are possible, arising from the increased availability of empirical data relating to the impacts of aggregate extraction activity, and of other sources of physical perturbations at the seabed. The relatively recent history of environmental assessments at aggregate extraction sites is an important limiting factor both in the generation of plausible ‘effects’ hypotheses, and in their subsequent validation. In particular, the paucity of longer time-series data for coarser substrata generated by consistent sampling practices is problematic (see 5. below).

Measures, which illuminate purely physical impacts on the benthic habitat, may present fewer conceptual difficulties and may, therefore, have greater short-term potential for application.

Another important consideration relating to the outcome of field evaluations is that, except in the case of gross environmental impacts, unqualified attribution of cause/effect is rarely possible. Accordingly, judgements are required on the *balance of probability* that an effect is attributable to a particular cause.

4. Responsive in a predictable and, preferably, linear way to variation in impact

Predictability in the behaviour of a measure in response to variability in impact from a specific source will be an important attribute of any EcoQO. Practical constraints for consideration will include changes in dredging practices in the course of a monitoring programme, the influence of other man-made activities in the locality, and extreme natural events, all of which may confound the interpretation of changes in a quality index. Also, regional variations in manifestations of impacts (*i.e.* in nature or severity) may arise from the influence of different hydrographic regimes or biogeographical factors. Thus an important issue for Workshop consideration will be the scope for, and desirability of, deriving site-specific measures of environmental impacts. Finally, evidence from studies of other man-made impacts suggests that, at least for the biota, changes are more likely to conform to cyclical (successional) rather than linear models.

5. Insensitive to natural change

Although, in practice, this represents another unattainable goal in relation to the determination of physical impacts, it may be feasible to define boundaries for expected natural changes, which would then constitute a basis for EcoQO derivation. An alternative approach may involve the paired sampling of environmentally similar 'treatment and 'reference' sites in such a way that the occurrence of synchronous natural changes may be cancelled out. Both these approaches, which incorporate a *temporal* element to the evaluation of change, have been employed in impact assessments, and merit Workshop consideration. Associated issues for attention include:

- the importance of effective site selection
- sampling frequency and intensity to meet statistical requirements
- appropriate statistical methodology
- site-specificity in the derivation of appropriate measures
- the importance of time-series data for retrospective testing of the behaviour of an index prior to its routine application

6. Relevant (if not directly, then by quantifiable association with matters of real concern such as population or community health, habitat or wider ecosystem integrity, and the performance of commercial fisheries)

The quantitative expression of relevance of an environmental measure is rarely attainable on local scales, given financial constraints on the scope of most environmental surveys. However, the substance of the title usefully serves to set a 'benchmark' against which existing measures may be evaluated. In practice, certain measures may allow plausible inferences concerning the possibility of wider consequences, even if the link cannot be readily quantified. Other measures may be justified on the grounds that they act as reliable indicators of the impacts of dredging activity, irrespective of their wider relevance. (This may be a practical necessity in view of the cost of an ecosystem-level study). Such measures may be used to justify management action on the basis of a precautionary approach, especially if studies elsewhere provide supporting evidence for the possibility that wider consequences might ensue.

The attribution of 'relevance' to dredging-induced changes, and the capacity of measures to reflect this, is clearly an important issue for Workshop attention, since EcoQOs are intended to be the tools by which an ecosystem approach to environmental management is implemented. In particular, the potential role of ecosystem-level modelling as a tool in impact assessment, and in guiding the selection of appropriate measures, merits consideration.

7. ***Defensible in a court of law, and therefore suitable for use as a statutory enforcement tool in regulating the environmental consequences of extraction activity***

As yet, there is no precedent for the adoption of an EcoQO approach in evaluations of the environmental effects of marine aggregate extraction and it is anticipated that, in any future applications, EcoQOs would initially be employed as 'triggers' or 'action levels' for confirmatory work. The ultimate test of the utility of an EcoQO (along with that of its originator) might be viewed in terms of its capacity to survive a process of legal interrogation. This is not included as a major discussion point, but rather as a sobering reminder of the importance of scientific rigour as an underpinning to the employment of measures for marine environmental protection.

Conclusions

Conclusions from this review are that:

- EcoQO derivation must be founded on good science and, at the same time, must be meaningful and communicable to non-specialists
- the scientific need is likely to be satisfied by the employment of complementary measures
- the statistical requirements of compliance monitoring, along with issues of cost-effectiveness, will also influence the choice of measures and associated activity, including sampling design and sampling practices, which may vary from site to site
- all aspects relating to application of an EcoQO approach must be subject to well-defined and rigorous QA/AQC procedures
- the paucity of relevant impact studies and time-series data for coarse-ground habitats limits the present capacity to derive and evaluate prospective EcoQOs
- caution should be exercised in initial application of an EcoQO approach, and some allowance made for adaptation in the light of practical experience
- close attention should be paid to the implications for future EcoQO derivation of developments in acoustic techniques for seabed discrimination and the application of ecosystem models.

Towards an understanding of the impacts of marine aggregate extraction on the seabed as a basis for developing an EcoQO approach for management.

S. E. Boyd and H. L. Rees
Centre for Environment, Fisheries and Aquaculture Science
Burnham-on-Crouch, Essex, CM0 8QD
E-mail: S.E.Boyd@cefas.co.uk and H.L.Rees@cefas.co.uk

Introduction

Industrial exploitation of the UK marine aggregate resource peaked in 1989, and has remained relatively steady in recent years at around 23 million tonnes *p.a.* from around the England and Wales coastline (Crown Estate Commission records). Increasing constraints on land-based exploitation and recognition that, against an estimated workable UK marine resource of about 1 billion tonnes (Crown Estate Commission records), suggests controlled extraction may be sustainable for the foreseeable future. Concerns over the effects of marine sand and gravel extraction on the environment and fisheries have grown over time, and this is particularly the case at localities off the eastern and southern English coastlines which are characterized by the occurrence of multiple dredging licences in close proximity.

Issues such as the potential for conflict of interest between stakeholders in the resource, and the efficacy of remedial measures during and after extraction, are analogous with land-based activities. However, in the marine environment, their resolution is rendered more difficult on account of the relative inaccessibility of sites, the 'open' nature of marine ecosystems relative to licensed boundaries, the general paucity of site-specific data on the structure and functional role of the habitat and biota associated with commercially exploitable reserves, and problems in quantifying the performance of local fisheries. Further momentum to concerns over environmental impacts is provided by developing interests in the conservation of marine biodiversity following the Rio Earth Summit, and in the protection of marine habitats (under the EU 'Habitats' Directive) and of whole sea areas, through international management initiatives under OSPAR and the EU. From such initiatives, concepts such as the 'ecosystem approach' to environmental management were prompted and this has led to consideration of the development of Ecological Quality Objectives (EcoQO's) [see for example, Lanters *et al*., 1999 for definition of terms]. As a tool in the management and regulation of marine aggregate extraction the setting of EcoQO's has obvious appeal, in that it offers the potential to improve transparency and promote responsibility. However, in deriving EcoQO's, there needs to be a sound scientific rationale, a full understanding of cause and effect relationships, and a clear link between the management measures and the desired level of ecological quality (see for example, Frid and Hall, 2001; Rees, 2001).

This paper briefly reviews the current state of knowledge relating to the impacts of marine aggregate extraction on the sediments and associated biota, and considers areas for further research in developing an EcoQO's approach for the management of dredging operations.

Methods of Dredging

Typically, marine aggregate in UK waters is dredged by trailer suction hopper dredgers. These employ a single, rear-facing pipe with centrifugal pump mounted either within the hull of the vessel or on the dredge pipe itself. Dredging using trailer suction hopper dredgers is carried out whilst the ship is underway leading to the production of shallow linear furrows approximately 1 to 3m in width and generally 0.2 to 0.3m in depth. Repeated dredging by trailer dredgers can result in substantial lowering of the seabed across a wide area and this will be related to the frequency of dredging and the level of dredging intensity (Norden Andersen *et al*., 1991). Whilst the main method of dredging in the U.K. is through trailing dredging, a number of vessels in the UK fleet are also able to dredge by anchoring or remaining stationary over the deposit. This is usually referred to as static dredging and is employed in areas where the deposit is spatially restricted or locally thick (e.g. East of the IOW and in the Bristol Channel). In this case, dredging usually results in saucer shaped depressions up to 8 to 10m deep, 200m in diameter and with slopes of <5%.

Impacts of Marine Aggregate Extraction

The environmental impacts of dredging have been well documented, with general reviews of the topic provided by Johnston (1981), Hurme and Pullen (1988), de Groot (1979a, 1979b; 1986), Gayman (1978), ICES (1992), Kenny (1995), Newell *et al*. (1998), ICES (*in press*) and Leuchs and Nehring (*unpublished*). From these reviews, it is clear that most studies have been concerned with impacts from maintenance dredging and beach recharge projects. Studies of the consequences of marine aggregate extraction on the seabed, in the U.K. and elsewhere are more limited, and largely confined to experimental circumstances (e.g. Shelton and Rolfe, 1972; Cressard, 1975; Millner *et al*., 1977; van Moorsel and Waardenburg, 1990, 1991; van Moorsel, 1993, 1994; Desprez and Duhamel, 1993; Kenny and Rees, 1994; 1996; Kenny *et al*., 1998; van Dalfsen and Essink, 1998; Jewett *et al*., 1999; Desprez, 2000). Investigations of the physical and biological status of licensed areas in the U.K. at various times following cessation of commercial dredging are very limited (e.g. Millner *et al*., 1977; Boyd *et al*., 2001) and so judgements as to the likely progress towards environmental restoration and the time-scales involved continue to be based on predictions rather than real data.

The Nature of Physical Effects

The length of time that trailer-dredged furrows or depressions created by static dredging will remain as distinctive features on the seabed depends on the ability of tidal currents or wave action to move sediments into them (van der Veer *et al*., 1985; Dickson and Lee, 1972; Millner *et al*., 1977; McGrorty and Reading, 1984). Erosion of dredge tracks in areas of moderate wave exposure and tidal currents have been observed to take between 3 and 7 years (Millner *et al*., 1977; Kenny and Rees, 1996; Essink, 1997; Boyd *et al*., 2001). At an experimental dredged site off Norfolk in 25m of water, dredge tracks were documented as being completely eroded well within 3 years of the cessation of dredging (Kenny and Rees, 1994; 1996; Kenny *et al*., 1998). In this case infill resulted from mainly sand in transport. However, in an area exposed to long period waves, dredge tracks (of 0.3-0.5m deep) were found to completely disappear in a gravelly substrate in 38 meters of water within 8 months (van Moorsel and Waardenburg, 1991; van Moorsel, 1993; 1994). In contrast, dredged depressions created by static dredging have been reported to remain as recognizable seabed features for a considerable time at Hastings (Shelton and Rolfe, 1973). Indeed, Dickson and Lee (1973) concluded that many years, perhaps amounting to decades, would be required for the dredged seabed to revert to its pre-dredging condition.

Changes in sediment composition as a result of dredging are well documented in the literature (Dickson and Lee, 1972; Shelton and Rolfe, 1972; Kaplan *et al*., 1975; Jones and Candy, 1981; Jones, 1981; Bonsdorff, 1983; van der Veer *et al*., 1985; Desprez, 1995; Kenny and Rees, 1994; 1996; Essink, 1997; Kenny *et al*., 1998; Jewett *et al*., 1999). Such changes range from minor alterations to surficial granulometry (McCauley *et al.*, 1977; Poiner and Kennedy, 1984) to an increase in fines (Jones and Candy, 1981) or decrease in fines (López-Jamar and Mejuto, 1988).

As infill of dredged depressions or tracks is typically dependent on the mobilization of fine material by tidal currents, this can result in a change of sediment composition from an admixture of sand and gravel to finer deposits (Dickson and Lee, 1972, 1973; Shelton and Rolfe, 1972; van der Veer *et al*., 1985; Essink, 1997). A change in the composition of sediment following aggregate extraction was observed in intensively dredged sediments off Dieppe (Desprez and Duhamel, 1993). At this site, the fine material, which infilled the dredged furrows, resulted from a combination of plume settlement and from the transport and trapping of bedload sediments. Significant changes in bathymetry brought about by dredging also has the potential to cause a drop in current strength resulting in the deposition of finer sediments (Dickson and Lee, 1972, 1973; Shelton and Rolfe, 1972; van der Veer *et al*., 1985) which may contribute to a localised depletion of oxygen (Bonsdorff, 1983; Norden Andersen *et al*., 1991). In contrast, an experimental study off North Norfolk reported the coarsening of seabed sediments, as a result of the exposure of deeper gravelly deposits following dredging (Kenny and Rees, 1996). This effect is consistent with post-dredging changes reported elsewhere (Jewett *et al*., 1999).

Particular dredging practices can also contribute to the fining or coarsening of sediments over time. For example, the aggregate industry carries out screening activities in order to meet specific sand/gravel requirements of the construction industry. Typically the construction industry requires marine aggregate to be supplied with a gravel content of greater than 50% (Mark Russell, pers. comm.). Where the *in-situ* gravel content of the dredged resource is lower than this, dredgers employ on-board screening to increase the gravel content of cargoes. Vessels use either static screen boxes or screening towers to alter the composition of the dredged aggregate, 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 material and water will pass through the screen, and be returned to the sea by means of a reject chute. This process can be reversed if the intention is to produce a sand cargo, with the coarse fraction of the dredged aggregate being rejected. Over time, this screening activity has the potential to significantly change the composition of sediments within a dredged area.

Dredging can also lead to the production of plumes of suspended material. Material can arise from the mechanical disturbance of the seabed sediment by the draghead (Moran, 1991). However, the outwash of material from spillways from the vessel hopper can generate a far greater quantity of suspended material (Coastline Surveys Limited, 1998). A further source of suspended material results from the rejection of unwanted sediment fractions by screening activities. Suspended sediments arising from the latter two processes have been termed surface plumes (Hitchcock and Drucker, 1996). Their areal extent and excursion are dependent on the particle size and total quantity of material suspended and the local hydrodynamics (Hitchcock and Drucker, 1996; Coastline surveys, 1998). Recent work has suggested that the sedimentation of particulate matter is principally confined to a zone of a few hundred meters from the point of discharge via the spillways (Coastline Surveys, 1998). Nevertheless the significance of sedimentation from plume fall-out on the benthic fauna and its effect on the rate of recolonisation is unknown.

The nature of biological effects

The most significant consequence of marine aggregate extraction is the removal of the substrate and the associated benthic fauna (ICES, 1992). Most studies on the effects of aggregate extraction have concentrated on establishing the rates and processes of macrobenthic recolonisation upon cessation of dredging (Cressard, 1975; Bonsdorff 1980; 1983; Hily, 1983; van der Veer *et al*., 1985; van Moorsel and Waardenburg, 1990, 1991; van Moorsel, 1993, 1994; Desprez and Duhamel, 1993; Kenny and Rees, 1994; 1996; Jewett *et al*., 1999; Desprez, 2000). These studies indicate, typically, that dredging causes an initial reduction in the abundance, species diversity and biomass of the benthic community. In a controlled study of the impacts of marine gravel extraction on the macrobenthos at a site off the east coast of England, Kenny and Rees (1994) showed that significant reductions had occurred in numbers of species (62%), abundance (94%) and the biomass (90%) following the removal of 52,000t of material by a trailer suction dredger.

A number of factors will influence the effect of dredging on local populations and their ensuing restoration. These include the types of organisms which remain in the vicinity (Thrush *et al*., 1991, 1992) following sediment extraction, the life histories and mechanisms of dispersal in different fauna (Levin, 1984), the patchiness of the environment (Thistle, 1981; Hall *et al*., 1994), the spatial and temporal variability of dredging disturbance (Hall *et al*., 1994), the effects of existing or new residents on the substratum (Dean and Hurd, 1980; Rhoads and Boyer, 1982; Davoult and Richard, 1990), and the potential interaction between dredging disturbance and other perturbations (e.g. the interaction between effects of mining and storm disturbance: Jewett *et al*., 1999).

Assessments of ‘recovery’ usually involve an examination of a number of community parameters such as abundance, numbers of species, diversity and biomass prior to disturbance and then at various intervals subsequently (Bonsdorff, 1980, 1983; Hily, 1983; Bonvicini Pagliai *et al*., 1985; van Moorsel and Waardenburg, 1990, 1991; van Moorsel, 1993, 1994; Kenny and Rees, 1994, 1996; Kenny *et al*., 1998; van Dalfsen and Essink, 1998; Jewett *et al*., 1999). Such studies also usually involve the monitoring of one or more reference areas (Stickney and Perlmutter, 1975; Bonsdorff, 1980; Bonvicini Pagliai *et al*., 1985; van Moorsel and Waardenburg, 1990,1991; Desprez and Duhamel, 1993; van Moorsel, 1993; 1994; Kenny and Rees, 1994, 1996; Kenny *et al*., 1998). The degree of 'recovery' can therefore be assessed by comparison of various measures following the design strategy known as Before After Control Impact (e.g. the B.A.C.I. approach) [Skalski and Mackenzie, 1982]. This design has been employed with considerable success by sampling dredged ('treatment') and non-dredged ('reference') sites to assess the rates of macrobenthic 'recovery' (Van Dolah *et al*., 1991; van Moorsel and Waardenburg, 1990, 1991; van Moorsel, 1993, 1994; Kenny and Rees, 1994; 1996; Kenny *et al*., 1998; Jewett *et al*., 1999). Community attributes that are most commonly investigated in this respect, include: numbers of species, abundance, biomass and the age structure of populations (Bonsdorff, 1983; Rees, 1987; van Moorsel and Waardenberg, 1991; van Moorsel, 1993; Desprez and Duhamel, 1993; Kenny and Rees, 1994; 1996; Kenny *et al.*, 1998; Essink, 1997, van Dalfsen and Essink, 1998; Jewett *et al*., 1999). Such comparisons are essential in establishing whether responses to aggregate extraction differ from prevailing community dynamics, and when (or if) the disturbed habitat begins to approximate ambient conditions. Typically, re-establishment of biomass dominants and age structures tends to take longer than other community attributes to return to pre-dredging levels following aggregate extraction (Bonsdorff, 1983; Rees, 1987; van Moorsel and Waardenberg, 1991; van Moorsel, 1993; Deprez and Duhamel, 1993; Kenny and Rees, 1996; Kenny *et al.*, 1998; Essink, 1997; van Dalfsen and Essink, 1998).

Models of Response

From studies of dredged sites (Shelton and Rolfe, 1972; Kaplan *et al*., 1975; Cressard, 1975; Bouchot *et al*., 1975; Hily, 1983; van Moorsel and Waardenberg, 1991; Desprez and Duhamel, 1993; van Moorsel, 1993,1994; Kenny and Rees, 1994; 1996; Desprez, 1997; Kenny *et al*., 1998) and from observations following defaunation as a consequence of storm disturbance (Eagle, 1973; 1975; Rees *et al.*, 1977) a general pattern of recolonisation is emerging (see also ICES *in press*). The first stage involves the settlement of a few opportunistic species, which are able to take advantage of the dredged and sometimes unstable sediments (Hily, 1983; van Moorsel, 1994; Kenny and Rees, 1994, 1996; Kenny *et al*., 1998). In the case of the North Norfolk experiment, the initial phases of recolonisation were dominated by opportunistic species such as *Dendrodoa grossularia* and *Balanus crenatus* (Kenny and Rees, 1994, 1996; Kenny *et al*., 1998). At the Klaverbank, *Magelona papillicornis*, a polychaete species, which normally inhabits sandy sediments, quickly invaded sediments following gravel extraction. Recolonization can either be by adults or larvae from the surrounding area if the disturbed area is similar to the original substrate (Cressard, 1975) or by larvae from more distant sources if the sediment is markedly different (Hily, 1983; Santos and Simon, 1980). These species can substantially increase the overall abundance and the numbers of species during the early stages of post-dredging recolonisation (Hily, 1983; van Moorsel and Waardenberg, 1991; van Moorsel, 1993; 1994; Kenny and Rees, 1994; 1996; Desprez, 1997; Kenny *et al*., 1998). A second phase is characterised by a reduced community biomass (van Moorsel and Waardenberg, 1991; van Moorsel, 1993; 1994; Desprez, 1997) which in North Norfolk persisted for three years (Kenny and Rees, 1994; 1996; Kenny *et al*., 1998). This is a natural expectation that biomass will remain reduced, while new colonisers 'grow on' to maturity comparable with the pre-dredging age/size profile. It has also been suggested that a reduced biomass is caused by increased sediment (mainly sand) in transport which scours the epibenthos. Paradoxically, it is this sediment that is also responsible for the infilling of dredge tracks (Kenny *et al*., 1998; ICES *in press*). Over time, the bedload transport approaches the pre-dredged equilibrium, allowing the restoration of community biomass (Kenny *et al.*, 1998). However in cases, where sediments are regularly disturbed by tidal currents, such as areas off Lowestoft, the community may be maintained at an early successional stage (Kenny and Rees, 1994). A similar model of response has been represented schematically by Hily (1983) and includes a further stage in which opportunists are replaced by a greater number of species. It was suggested that this replacement was the result of increasing levels of interspecific competition. However, this model was based on observations following the dredging of a sandy mud (Hily, 1983), thus further evidence is required to establish whether such oscillations occur in more stable gravel habitats during the latter stages of succession. The challenge now in moving towards an EcoQO approach is to validate and or refine this and other relevant models of response and to start to numerically ascribe the degree of change, for example, in terms of the numbers and densities of species (Elliot, 1996).

Rates of macrofaunal recovery

The estimated time required for re-establishment of the benthic fauna following marine aggregate extraction may vary depending on the nature of the habitat, the scale and duration of disturbance, hydrodynamics and associated bed-load transport processes, the topography of the area and the degree of similarity of the habitat with that which existed prior to dredging (for review see Newell *et al.*, 1998). Available evidence, largely obtained from experimental studies, suggests that substantial progress towards 'recovery' could be expected within 2-3 years of cessation of dredging in sandy gravel habitats exposed to moderate wave exposure and tidal currents (de Groot, 1979; van Moorsel, 1993; Desprez and Duhamel, 1993;van Dalfsen and Essink, 1998; Kenny *et al.*, 1998; Newell *et al.*, 1998; Desprez, 2000). However, preliminary observations from a recent study of a historic commercial extraction site off Harwich indicate that the 'recovery' period may be more prolonged (i.e. > 4years), especially for sites dredged repeatedly (Boyd *et al.*, 2001).

A Framework for Future Studies

As yet, co-ordinated studies on a wide geographical scale investigating the physical and biological recovery of commercial aggregate extraction sites in the U.K. are limited, although one study funded by DTLR, DEFRA and CEC and a parallel investigation initiated by BMAPA are addressing this issue. Based on existing evidence, however, the two most commonly encountered scenarios in the aftermath of marine aggregate extraction are:

a) sites where the substratum has changed from a sandy gravel to a gravelly sand;
b) sites where the substratum has remained unchanged.

This is not to exclude the possibility of other consequences such as the exposure of clay depending on local circumstances. In the first of the scenarios, alterations to the sediment could result in a number of ways, including the exposure of an underlayer of finer sediments, discharge of finer sediments from spillways or screening, and the trapping of bedload in dredged furrows. The degree of change will depend on the local circumstances, but it is also likely to be related to the magnitude of perturbation. It is postulated that the colonising benthic fauna will reflect this change to the substrate, through a shift in the proportions of sandy *versus* gravelly fauna. Accompanying this it is also postulated that there would be a net decline in biomass. This model of response is schematically portrayed in Figure 5.

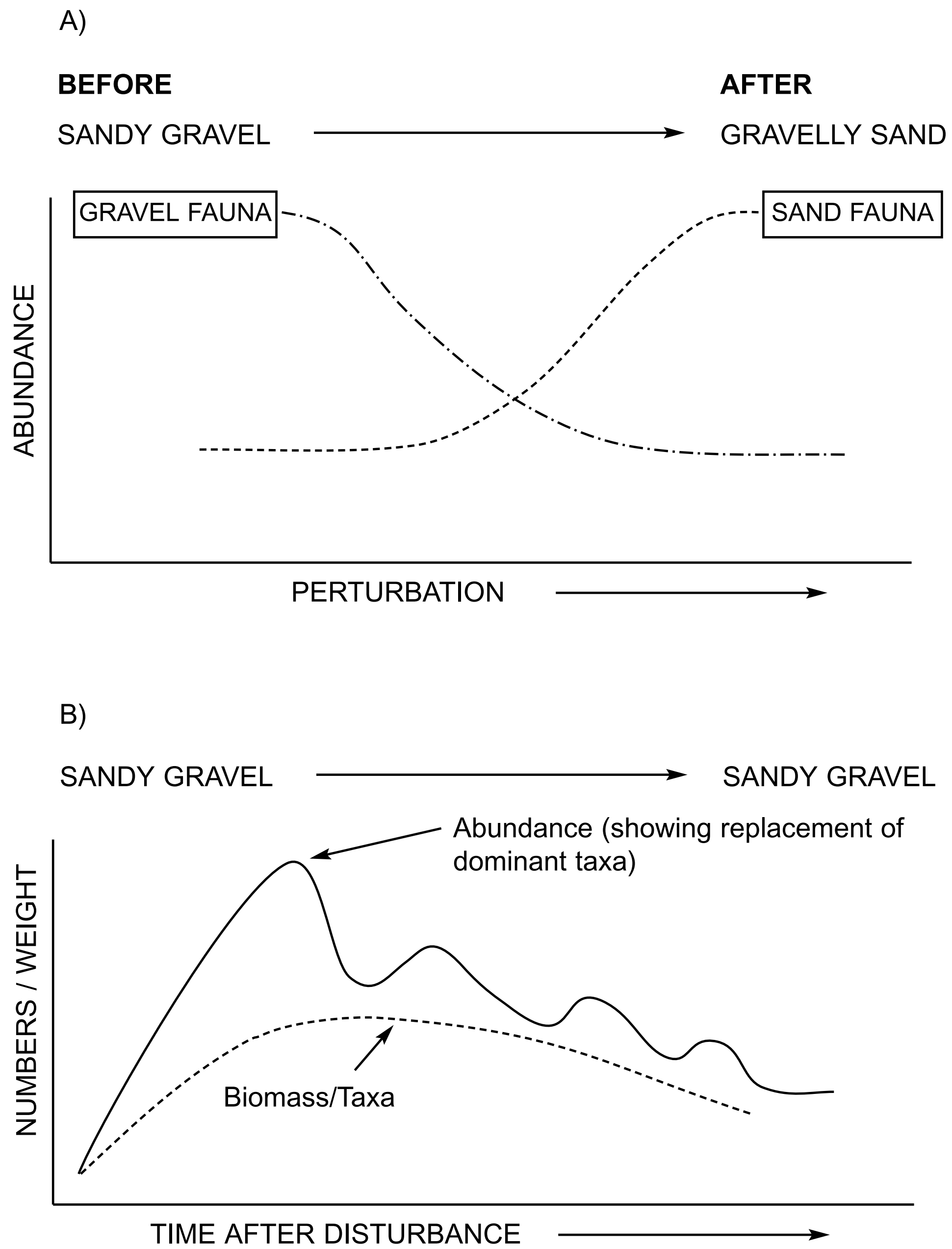

Figure 5. A) Simplified diagram of changes in the proportions of gravelly fauna in response to a change in sediment type as a consequence of marine aggregate extraction. B) Simplified model of changes in the benthos after the cessation of marine aggregate extraction.

In the second scenario, sediments in the aftermath of aggregate extraction are similar to that which existed prior to disturbance i.e. sandy gravels. Thus it is predicted that the fauna recolonising such sites will follow classical successional dynamics. It is acknowledged that there are numerous factors and processes involved in the physical and biological recovery of dredged sediments and some of these have been discussed earlier. The validity of these simplified models remains to be tested, and accumulated field data will doubtless lead to some modifications. However, they provide a useful framework for the selection of trial locations, the evolution of sampling strategies and their time-scales, in order to evaluate post cessation recolonisation and recovery rates and, eventually, to provide a reliable predictive capability that can be used to set EcoQO's.

Summary

Many of the field studies reported in the literature are the results of controlled field experiments and these have proved useful in determining the rates and processes leading to benthic re-establishment following aggregate extraction. From such studies, a general pattern of response to marine aggregate extraction is emerging and has been described. This needs to be tested to establish its general validity in all environments particularly in areas which have been exposed to commercial dredging practices. From such work, it is clear that re-establishment of a community similar to that which existed prior to dredging can only be attained if the topography and original sediment composition are restored (for a contrary view see Seiderer and Newell, 2000). Should physical stability of the sediments not be attained, however, it has been postulated that communities will remain at an early developmental stage (Kenny and Rees, 1996; Kenny *et al*., 1998, Boyd *et al*., 2001).

Processes affecting settlement and recruitment to local populations are well represented in the literature, although most studies relate to experience of working in soft-sediments (e.g. Santos and Simon, 1980; Smith and Brumsickle, 1989; Günther, 1992; Whitlatch *et al*., 1998). A number of key research areas therefore need to be developed to provide empirically-grounded models for physical and biological responses, with a predictive element on the effects of commercial aggregate extraction. Such predictive models are essential for deriving scientifically robust EcoQO's.

Despite the relatively straightforward rationale for quantitative predictions (see for example Underwood, 1990), the models to underpin these predictions may be difficult to formulate. This is because the results from different studies may be incomparable or be insufficient to account for all the various dredging scenarios, which have occurred or are currently practised in UK waters. This does not make the task impossible. The challenge is to devise robust sampling strategies for a suite of locations representative of different effective dates of cessation of dredging for follow-up field sampling. These extraction sites should, as far as possible, be selected to represent a range of geographical conditions around the coastline. In parallel, strategies need to be devised for enhancing comparisons between studies and for synthesising existing data suitable for the development of generic models of response.

Many of the reported experimental studies have had the advantage of referring to a baseline survey of the dredged area prior to the commencement of dredging. In addition such studies have often sought reference areas exposed to similar environmental conditions to the dredged site which can be monitored over time to track natural environmental fluctuations. This strategy is compatible with an EcoQO approach which attempts to monitor attributes of ecological quality (EcoQ), relative to a reference level and against an *a priori* derived standard (Elliot, 1996; Lanters *et al*., 1999). However, baseline information in most cases is unavailable for commercial extraction sites in U.K. waters and after dredging has taken place for many years, the benthos (and sediments) may have been structurally altered as a consequence of dredging (Dickson and Lee, 1972; Shelton and Rolfe, 1972; Kaplan *et al*., 1975; Jones and Candy 1981; Jones, 1981; Bonsdorff, 1983; van der Veer *et al*., 1985; Desprez, 1995; Kenny and Rees, 1994; 1996; Essink, 1997; Kenny *et al*., 1998; Jewett *et al*., 1999). In this situation, it is difficult to reach a judgement as to what constitutes an appropriate reference level of a particular environmental indicator and whether the level is representative of the likely pre-dredged status. Therefore, in carrying out studies in 'commercial' deposits, investigators will be restricted to documenting patterns of community change or comparing the fauna of areas presumed to represent different 'recovery' stages and inferring underlying causal mechanisms. Nevertheless studies at extraction sites have the unarguable advantage of providing statements on the reality of conditions in the aftermath of actual commercial exploitation. A body of case studies on the consequences of marine aggregate extraction over sufficiently long time-scales is therefore required to underpin the derivation of reliable and scientifically credible EcoQO's.

References

BONSDORFF, E., 1980. Macrozoobenthic recolonisation of a dredged brackish water bay in SW Finland. Ophelia, *1*:145-155.

BONSDORFF, E., 1983. Recovery potential of macrozoobenthos from dredging in shallow brackish waters. *Oceanologia acta* Auswirkungen von baggern und Verklappen auf die Fauna - Literaturstudie. Dommermuth, M. Koblenz, Stand, Bundesanstalt für Gewässerkunde.

LEVIN, L. A., 1984. Life history and dispersal patterns in a dense infaunal polychaete assemblage: community structure and response to disturbance. Ecology *65*: 1185-200.

LÓPEZ-JAMAR, E. AND MEJUTO, J., 1988. Infaunal benthic recolonisation after dredging operations in La Coruna bay, NW Spain. SP: 27-32.

BONVICINI PAGLIAI, A. M., COGNETTI VARRIALE, A. M., CREMA, R., CURINI GALLETTE, M., AND VANDINI ZUNARELLI, R., 1985. Environmental impact of extensive dredging in a coastal marine area. Marine Pollution Bulletin, *16*:483-488.

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

BOYD, S.E., LIMPENNY, D. S., REES, H.L. AND COOPER, K.C., 2001. Preliminary observations of the effects of dredging intensity on the recolonisation of dredged sediments (Area 222). In: Assessment of the re-habilitation of the seabed following marine aggregate dredging. 1st Annual report to the DETR Contract MPO676.

CRESSARD, A., 1975. The effects of offshore and gravel mining on the marine environment. Terra et Aqua *8/9:* 24-33.

DAVOULT, D. AND RICHARD, A., 1990. Experimental study of the recruitment of the sessile community on the pebbly sea bed in the Dover Strait. Cah.Biol.Mar. *31*:181-199.

DE GROOT, S. J., 1979a. 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.

DE GROOT, S. J., 1979b. The potential environmental impact of marine gravel extraction in the North Sea. Ocean Management *5*:233-249.

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.

DESPREZ, M., 2000. Sedimentological and biological effects of sand deposition around the dredging site of Dieppe (F). ICES CM 2000/E:07 8pp.

DESPREZ, M. AND DUHAMEL, S., 1993. Comparison of the impact of gravel extraction on geomorphology, sediment & macrofauna in two areas: Klaverbank (NL) & Dieppe (F). ICES C.M.1993/E:7:51-66.

DICKSON, R. R. AND LEE, A., 1972. Study of effects of marine gravel extraction on the topography of the seabed. CM 1972/E:25, 1-19.

DICKSON, R. R. AND LEE, A., 1973. Gravel extraction: effects on seabed topography. Offshore services *6*:32-36.

EAGLE, R. A., 1973. Benthic studies in the south-east of Liverpool Bay. Estuarine and Coastal Marine Science, *1*:285-299.

EAGLE, R. A., 1975. Natural Fluctuations in a soft-bottom benthic community. Journal of the Marine Biological Association of the United Kingdom *55*: 865-878.

ELLIOT, M, 1996. The derivation and value of ecological quality standards and objectives. Marine Pollution Bulletin *32*(11): 762-763.

ESSINK, K., 1997. Risk analysis of coastal nourishment techniques (RIACON) Final evaluation report. RIKZ-97.031, pp. 1-42. The Netherlands, National Institute for Coastal and Marine Management/RIKZ.

FRID, C. AND HALL, S., 2001. Ecological Quality Objectives for benthic communities: If we protect the habitat do we need to do more? ICES CM 2001/T:06.

GAYMAN, W., 1978. Offshore dredging study: Environmental ecological report. Ocean Management *4*:51-104.

HALL, S. J., RAFFAELLI, D. AND THRUSH S.F., 1994. Patchiness and disturbance in shallow water benthic assemblages. In Aquatic Ecology : Scale , Pattern and Process (ed. P.S. Giller , A.G. Hildrew and D.G. Raffaelli), pp. 333-75. Blackwell Science, Oxford.

HILY, C., 1983. Macrozoobenthic recolonisation after dredging in a sandy mud area of the Bay of Brest enriched by organic matter. Proceedings of the 17th European Symposium on Marine Biology, Brest (France). (Ed by Cabioch, L., Glemarec,-M., Samian,-J.F.) pp.113-120.

HITCHCOCK, D. R. AND DRUCKER, B. R., 1996. Investigation of benthic and surface plumes associated with marine aggregates mining in the United Kingdom. 2. The global ocean - towards operational oceanography. Oceanology International 1996.

HURME, A. K. AND PULLEN, E. J., 1988. Biological effects of marine sand mining and fill placement for beach replenishment: Lessons for other uses. Marine Mining *7*:123-136.

ICES, 1992. ICES co-operative research report No. 182. Report of the ICES Working Group on the effects of extraction of marine sediments on fisheries. 78pp.

ICES (*in press*). ICES co-operative research report. Report of the ICES Working Group on the effects of extraction of marine sediments on the marine ecosystem.

JEWETT, S. C., FEDER, H. M., AND BLANCHARD, A., 1999. Assessment of the benthic environment following offshore placer gold mining in the north-eastern Bering Sea. Marine Environmental Research *48*:91-122.

JONES, G. AND CANDY, S., 1981. Effects of dredging on the macrobenthic infauna of Botany Bay. Australian Journal of Marine Freshwater Research *32*:379-398.

KAPLAN, E. H., WELKER, J. R., KRAUS, M. G., AND MCCOURT, S.. 1975. Some factors affecting the colonization of a dredged channel. Marine Biology *32*:193-204.

KENNY, A. J., 1995. The biology of marine gravel deposits and the effects of commercial dredging. Unpublished Ph.D. thesis, University of East Anglia, 243pp.

KENNY, A. J. AND REES, H. L., 1994. The effects of marine gravel extraction on the macrobenthos: early post dredging recolonisation*.* Marine Pollution Bulletin *28*(7)*: 1*-19.

KENNY, A. J. AND REES, H. L., 1996. The effects of marine gravel extraction on the macrobenthos: Results 2 years post-dredging. Marine Pollution Bulletin *32*(8/9): 615-622.

KENNY, A. J., REES, H. L., GREENING, J., AND CAMPBELL, S., 1998. The effects of gravel extraction on the macrobenthos at an experimental dredge site off North Norfolk, UK (results 3 years post-dredging). ICES CM1998/V:14, 1-7.

LEUCHS, H. AND NEHRING, S. (unpublished). Cah.Biol.Mar. *29*:37-49.

MCCAULEY, J. E., PARR, R. A., AND HANCOCK, D. R., 1977. Benthic infauna and maintenance dredging: A case study. WR *11*:233-242.

MCGRORTY, S. AND READING, C. J., 1984. The rate of infill and colonization by invertebrates of borrow pits in the Wash (S.E. England). Estuarine, Coastal and Shelf Science *19*:303-319.

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.

MORAN, P.J., 1991. The effects of dredging on the larval settlement and community development of fouling on Port Kembla Harbour, Australia. Water Research *25*(9): 1151-1155.

NEWELL, R. C., SEIDERER, L. J., AND HITCHCOCK, D. R., 1998. The impact of dredging works in coastal waters: A review of the sensitivity to disturbance and subsequent recovery of biological resources on the seabed. Oceanography and Marine Biology *36*: 127-178.

NORDEN ANDERSEN, O. G. N., NIELSEN, P. E., AND LETH, J., 1992. Effects of seabed, benthic fauna and hydrography of sand dredging in Køge Bay, Denmark. 1-6. Helsingor, Denmark. Proceedings of the 12th Baltic Marine Biologists Symposium. Bjornestad, E., Hagerman, L., and Jensen, K.

POINER, R. AND KENNEDY, R., 1984. Complex patterns of change in the macrobenthos of a large sandbank following dredging. Marine Biology *78*:335-352.

REES, E. I. S., NICHOLAIDOU, A., AND LASKARIDOU, P., 1977. The effects of storms on the dynamics of shallow water benthic associations. In: Biology of benthic organisms, Anonymous pp. 465-474. Pergamon Press, Oxford.

REES, H. L., 1987. A survey of the benthic fauna inhabiting gravel deposits off Hastings, Southern England. 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. Background paper for this Workshop.

RHOADS, D. C., ALLEN, R. C., AND GOLDHABER, M. B., 1977. The influence of colonizing benthos on physical properties and chemical diagenesis of the estuarine sea floor. In: Ecology of Marine Benthos, edited by B.C.Coull, pp. 113-138. University of South Carolina Press, Columbia.

RHOADS, D. C., MCCALL, P. L., AND YINGST, J. Y., 1978. Disturbance and production on the estuarine seafloor. American Scientist *66*:577-586.

RHOADS, D.C. AND BOYER L.F., 1982. The effects of marine benthos on physical properties of sediments: a successional perspective. In: Animal- Sediment Relations (ed. P.L. McCall and M.J.S Tevez), pp.3-52 Plenum Press, New York.

SEIDERER, L. J. AND NEWELL, R. C., 2000. Analysis of the relationship between sediment composition and benthic community structure in coastal deposits: Implications for marine aggregate dredging. ICES Journal of Marine Science *56*:757-765.

SHELTON, R. G. J AND ROLFE, M. S., 1972. The biological implications of aggregate extraction: recent studies in the English Channel. C.M.1972/E:26, 1-12.

SKALSKI, J.R. AND MACKENZIE, D. H., 1982. A design for aquatic monitoring programs. J. Environ. Management *14*: 237-251.

LANTERS, R.L.P, SKJOLDAL, H.R. AND T.T. NOJI (editors), 1999. Ecological Quality Objectives (EcoQOs) for the North Sea. Workshop report, Scheveningen - the Netherlands. 1-3 September 1999: 591.

THISTLE, D., 1981. Natural physical disturbances and communities of marine soft bottoms. Marine Ecology Progress Series *6*:223-228.

THRUSH, S. F., PRIDMORE, R. D., HEWITT, J. E. AND CUMMINGS V. J., 1991. Impact of ray feeding disturbances on sand flat macrobenthos: do communities dominated by polychaetes or shellfish respond differently? Marine Ecology Progress Series *69*:245-52.

THRUSH, S. F., PRIDMORE, R. D., HEWITT, J. E., AND CUMMINGS, V. J., 1992. Adult infauna as facilitators of colonization on intertidal sandflats. Journal of Experimental Marine Biology and Ecology *159*: 253-265.

UNDERWOOD A. J., 1990. Experiments in ecology and management: their logics, functions and interpretation. Aust. J. Ecol. *15*: 365-89.

VAN DALFSEN, J. A. AND ESSINK, K., 1997. Risk analysis of coastal nourishment techniques in the Netherlands National Evaluation Report (The Netherlands). RIKZ-97.022, 1-98. Haren, the Netherlands, National Institute for Coastal and Marine Management/RIKZ.

VAN DALFSEN, J. A. AND ESSINK, K., 1998. The effects on macrozoobenthos of subaqueous sand extraction north of the Island of Terschelling, the Netherlands. Working Document RIKZ/OS 98.607 1998. Presented at the ICES Working group on the effects of extraction of marine sediments on the marine ecosystem.

VAN DER VEER, H. W., BERGMAN, M. J. N., AND BEUKEMA, J. J., 1985. Dredging activities in the Dutch Wadden Sea: effects on macrobenthic infauna. Netherlands Journal of Sea Research *19*:183-190.

VAN DOLAH, R. F., CALDER, D. R., AND KNOTT, D. M., 1984. Effects of dredging and open water disposal on benthic macroinvertebrates in a South Carolina Estuary. Estuaries, *7*:28-37.

VAN MOORSEL, G. W. N. M AND WAARDENBURG, H. W., 1990. Impact of gravel extraction on geomorphology and the macrobenthic community of the Klaverbank (North Sea) in 1989. 7-53.

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. Bureau Waardenburg bv. pp.5-54.

VAN MOORSEL, G. W. N. M., 1993. Long term recovery of geomorphology and population development of large molluscs after gravel extraction at the Klaverbank (North Sea). *92*(*16)*: 1-41.

VAN MOORSEL, G. W. N. M., 1994. The Klaverbank (North Sea), geomorphology, macrobenthic ecology and the effect of gravel extraction. *94(*24): 1-65. Culemborg, The Netherlands, Bureau Waardenburg bv.

Applying the EcoQO concept to potential fisheries impacts of marine aggregate extraction: Monitoring the outcome

S.I.Rogers & M.D.Nicholson
Centre for Environment, Fisheries and Aquaculture Science
Pakefield Road, Lowestoft, Suffolk NR33 OHT
E-mails: S.I.Rogers@cefas.co.uk and M.D.Nicholson@cefas.co.uk

Summary

The development of Ecological Quality Objectives (EcoQOs) has arisen from the desire to develop an ‘ecosystem approach’ to environmental management. This paper describes one suitable Ecological Quality applicable to declining fish populations, and sets an appropriate Objective and management framework. The need to monitor in order to show the success, or otherwise, of management measures is often overlooked. We show how subsequent fisheries monitoring is most effective when underpinned by a predetermined decision-making procedure which is related to meaningful scientific outcomes. The decisions which follow correspond to appropriate precautionary hypotheses, and they lead to adequate levels of sampling, unambiguous conclusions, and an agreed basis for decisions.

Introduction

When considering applications for marine dredging, and particularly applications in areas that are important for fish spawning, migration routes, or as nursery and over-wintering grounds, it is Government policy to adopt a precautionary approach (DETR, 2001). Where there is evidence of a decline in fish population size, in parallel with the continued extraction of marine aggregate, one management strategy has been to restrict aggregate extraction, on the assumption that aggregate removal may be impeding the onshore migration of fish. However, for the strategy to be complete, this must be accompanied by clear objectives, and a monitoring programme which can evaluate the success, or otherwise, of the management action.

Important Ecological Qualities for fish populations are a measure of the population size, its trend and rate of change, and the assurance that current populations are sustainable. The most important management objective for species which are under threat is to prevent further decline, and then restore population size and spatial extent. So, for a fishery in decline, the objective could be to restore current catches to some *status quo* level. If this is achieved using a seasonal restriction on dredging activity, it will be necessary to review its effectiveness at regular intervals. There is no value in applying a mitigation measure that does not achieve its intended purpose. However, there are problems with such as approach. If, after implementation of the dredging restriction, the fish catches remain the same or continue their steady decline, fishermen may press for a longer restriction on the assumption that the original was not effective. On the other hand, the aggregate industry will want it removed on the grounds that it has been ineffective. If catches increase, it will be assumed that the restriction was effective. The fishermen may then argue for it to be sustained, since the inverse-relationship between dredging and catches has been demonstrated. The aggregate industry could argue for the dredging restriction to be lifted, since it is now unnecessary. The basis for these disputes could be removed if there was a clear understanding of the basis for future decisions, and if these were identified and agreed before the restriction and the monitoring begin.

Hypotheses

In statistical tests, the traditional null hypothesis is that there has been no effect (e.g. of dredging). So if after the dredging restriction was applied there was no significant difference between fish catches and the *status quo* catch, it may be because there really was no effect to be seen, or it may be because there was an effect but the test had insufficient statistical power to find it. These hypotheses are decidedly un-precautionary. The burden of proof is on the fishermen to show that catches are less than the *status quo*. If they fail, the *status quo* is assumed to apply. One route to a precautionary approach is to reverse the roles of these hypotheses and agree an acceptable level of impact. The null hypothesis is then that the impact is worse than this agreed level. If a small reduction in *status quo* catches (e.g. less than 20%) is acceptable, then the monitoring programme has to show that the catches are significantly better than this, i.e. they fall within zone A of Figure 6. The burden of proof has now been reversed, reflecting the precautionary assumption that dredging could be having an effect.

A similar set of hypotheses can be defined to test whether the management measures have failed to improve the depleted catch. The procedure is reversed. Suppose there is an agreed recovery in catch rates that is considered to be the minimum that is worthwhile (e.g. greater than 20%). If, at some future date, catches were significantly less than 20%, then we would conclude that the restriction had had no effect, i.e. fell in zone C of Figure 6. These hypotheses are again precautionary, and place the burden of proof on monitoring to demonstrate that the small change in catches was significantly less than 20%, and within zone C of Figure 6.

Linking Hypotheses to Decisions

So far, no actions or decisions have been associated with the various hypotheses described above. These will be based on the best scientific understanding of the fishery, and may call for specific actions, such as further monitoring or a review. For example, for those catch rates which fall within Zone A (fishery recovered), the dredging restriction may have been effective and should be maintained, but monitoring should eventually be reduced to a "watch-dog" level to protect against further reduction. Catch rates which fall consistently within Zone B suggest that the fishery may be recovering, and that monitoring should continue. In Zone C (no recovery) the dredging restriction has had no effect and other mitigation measures should be evaluated.

Simple Statistical Tests for Making Decisions

There are many statistical procedures that could be adopted to make the necessary decisions. For example, these could be based on the maximum or minimum catch in a five-year period, which might be meaningful in terms of the biological characteristics of the stock. A test could also be based on the average or median catch, and this may be less stringent than requiring that every annual index in a given period complies with some reference value. However, these issues are mostly about the choice of procedure. The essential difference with this approach is making these choices explicit, and to reverse the burden of proof from showing damage (unprecautionary) to showing acceptable damage (precautionary).

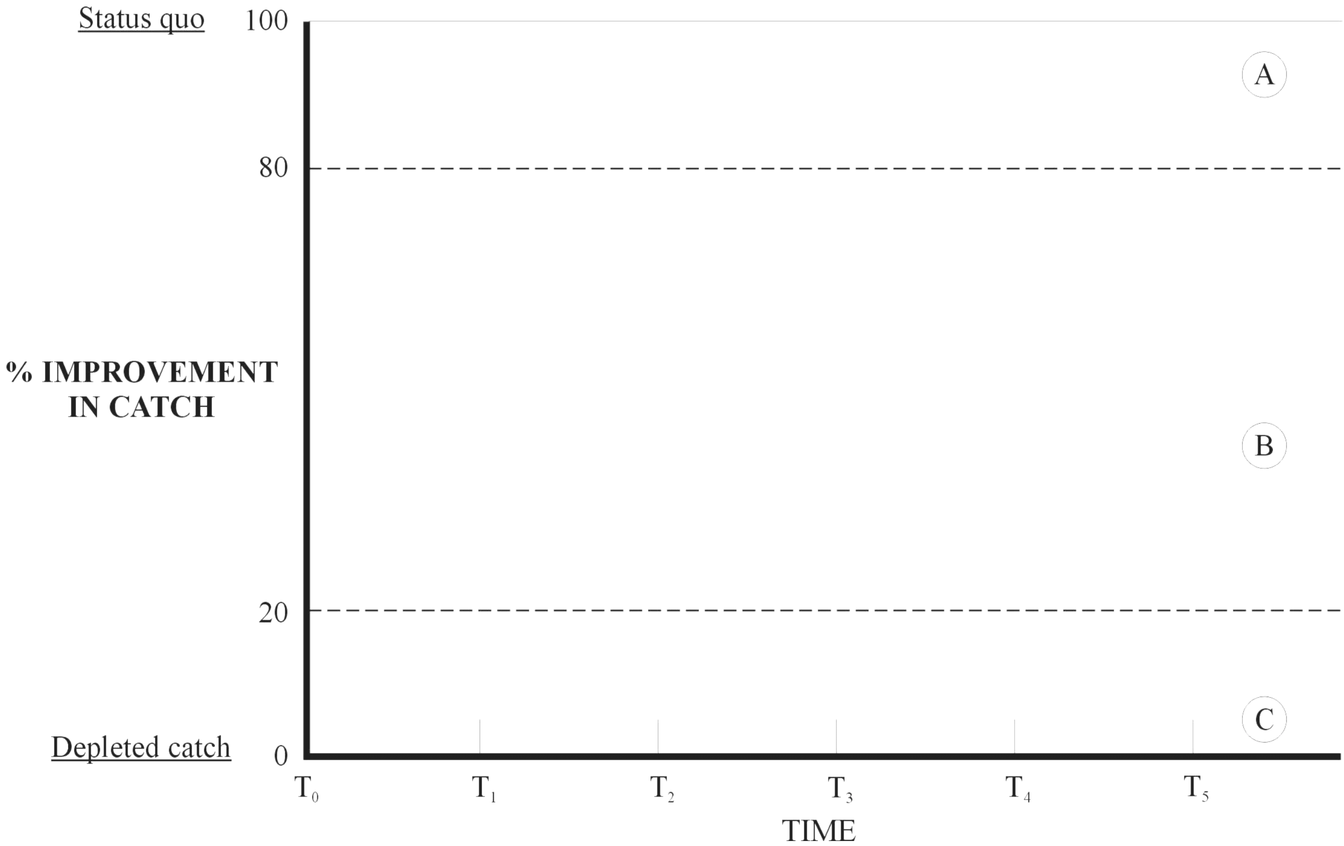

Figure 6. A schematic figure showing one method of evaluating precautionary hypotheses

Reference

DETR, 2001. Draft Marine Minerals Guidance Note 2: Guidance on the Extraction by Dredging of Sand, Gravel and other minerals from the English Seabed. Consultation Paper. Department of the Environment, Transport and the Regions, London. February 2001. 42pp.

3. WORKING GROUP OUTCOMES

WG 1. Develop EcoQOs (analogous to EQOs) taking account practicality, feasibility, regulatory interests, conservation and fisheries interests.

Chairman – Tom Simpson, DTLR
Rapporteur – Johanna Johnson, CEFAS

The Working Group discussed current marine aggregate extraction activities at length and the initiatives government and industry have taken to date to minimise deleterious impacts to the marine environment. The latter key issues are clearly set out in paragraphs 27 to 50 of Draft Marine Minerals Guidance Note 2 (MMG2, 2001) and are referred to under EcoQO 3 below.

Feasibility and Practicality of EcoQOs for Marine Aggregate Extraction

The Working Group spent some time discussing whether the EcoQO approach was appropriate for marine aggregate extraction (or other specific human activities) taking into account:

- ♦ the scale of the activity;
- ♦ its limited distribution;
- ♦ its often close proximity to other human activities and therefore the potential confusion of causes of impacts;
- ♦ the current state of knowledge of impacts of marine aggregate extraction;

It was implicit in the discussions that the main impacts from marine aggregate extraction are due to physical impacts and the consequential biological responses and that these are most significant at the seabed.

It was agreed that due to the potential interactions with other human activities, there would be problems with EcoQOs aimed specifically at marine aggregate extraction. However, it was agreed that metrics could be developed to assess marine aggregate extraction that would also be likely to be relevant to seabed disturbance in general. The Group took the view that these metrics (see the outcome of Working Groups 2 and 3) should fit within an overarching set of EcoQOs addressing marine aggregate extraction and that these Objectives may well be relevant to seabed impacts from other human activities as well. These overarching EcoQOs would be derived from the metrics.

The overall view was, therefore, that it was not practical or feasible to develop specific EcoQOs for marine aggregate extraction in isolation from EcoQOs for other human activities. However, it was agreed to be practical to develop metrics for marine aggregate extraction that contribute to overarching EcoQOs. At least some of those metrics would also be likely to be relevant to seabed disturbance by other human activities.

Current State of Scientific Knowledge in Relation to EcoQOs

The extent of current scientific knowledge was discussed and it was generally agreed that further research was needed to provide an appropriate understanding of the effects of marine aggregate extraction. It was generally felt that the results from ongoing studies would benefit the development of metrics for seabed disturbance and that these would contribute to the development of overarching EcoQOs.

Research Needs

The key research requirement identified was the assessment of the distribution of different habitat types and the identification of those which were considered to be the most important in terms of ecosystem structure and function, and therefore most in need of protection. It was agreed that continued investigation was needed on best practice for reducing the impacts of dredging, assessing habitat changes that occur post-dredging and the impact of noise from dredging.

Overarching EcoQOs

The Group then discussed potential overarching EcoQOs for marine aggregate extraction, and four were identified:

EcoQO 1: To have a proportion (x%) of each habitat that is protected from human activities.

- This objective would aim to ensure that an ecologically viable amount of each habitat, and the associated communities (biotopes), is protected in a near pristine condition.

- These areas will therefore provide a high degree of protection for relevant habitats and species, and they may act as sources of ‘seed’ larvae or mobile animals for recolonising impacted sites and replenishing their populations.

- The determination of the viable proportion (x%), and the spatial distribution, of the protected areas is the key research question that needs to be answered to inform society before decisions are made on how much to protect from human activities.

EcoQO 2: To ensure that the proportion of habitat and associated communities impacted does not prevent the proper functioning of that system during extraction and allows recovery once dredging ceases.

- Development of management practices will ensure impacts of dredging activities are minimised through reduction of dredged area, avoidance and protection of sensitive locations and use of best available techniques. See also EcoQO 3 below.

EcoQO 3: Incorporate best practice in dredging operations in order to promote the recovery of impacted ecosystems.

- In order to achieve this objective various conditions such as the following may be employed:
  - limiting working to discrete sub-areas within licensed areas i.e. zoning;
  - limits on extraction rates;
  - limits on the total quantity of material that can be removed;
  - restrictions on times when dredging is allowed;
  - restrictions on the type of dredger that operates;
  - the prohibition of screening at sea; and
  - a requirement to leave a substrate of similar characteristics to that which existed before dredging.

EcoQO 4: There should be no impact outside an agreed area of influence; this is the area of primary and secondary impacts. This will be defined and assessed as part of the Environmental Impact Assessment.

The area of primary impact is the area of seabed that the dredger's draghead directly comes into contact with as a consequence of the dredging process. The area of secondary impact is the area of sea that may be influenced by:

- the plume of sediment-laden water that overspills from the dredger while it is operating;
- noise from the dredger; and
- changes in currents and waves due to altered depth/topography resulting from the dredging.

Examples of secondary impacts are:

- increased suspended sediment in the water column;
- altered sediment deposition on the seabed;
- altered hydrodynamics; and
- noise.

It was recognised that the zone of secondary impact could extend outside the permitted area and that a key issue is therefore, how far outside the permitted area is acceptable? In discussion it was suggested that this would depend on the nature and sensitivity of the area that might be impacted, and the likely impacts.

WG 2. Define metrics appropriate for use as environmental indicators, and their state of development, practicality and feasibility:
Sub-group: Biological.

Chairman – Hubert Rees, CEFAS
Rapporteur – Sian Boyd, CEFAS

Scope of task

Experience accumulated over the past thirty years of marine environmental impact assessments has demonstrated that in most areas and circumstances one of the most responsive and easily addressed elements of the marine ecosystem following an imposed disturbance is the benthic macrofauna. As a result most marine monitoring and assessment studies have been based on recording changes in the distribution, density and community structure of this group. Consequently, this provided the focus for the sub-group discussions.

The range of targets, and the basis of potential standards, which might reasonably be developed into future EcoQOs, were discussed. It was recognised that there was potential for overlap between various initiatives towards the development of effective descriptors of ecological status, e.g. performance indicators, EQSs and EcoQOs/EcoQSs. In this respect, terminology was recognised to be problematic but, in the interests of time, the working group adopted a pragmatic approach, by concentrating on an evaluation of the *indicator potential* of a range of metrics which may be appropriate for use in an EcoQO context.

It was recognised that an EcoQO approach may serve different purposes. For example, one purpose may be to conserve threatened species and habitats, in which case structural integrity may be of primary importance, whereas the protection of a fisheries resource may be more concerned with ecosystem functioning, i.e. conceivably without regard to structural properties such as community diversity. It may be anticipated that the level of site-specific knowledge required for the development of EcoQOs will vary depending on the purpose and hence the target for investigation.

It was recognised that, ideally, EcoQOs would be responsive solely to the impact under study. However, in reality it is difficult to define an EcoQO (*based on current knowledge*) which is indicative of the impacts of aggregate extraction but which is not affected by other influences, especially natural events. Thus the importance of identifying metrics which, as far as possible, permitted the elucidation of unambiguous cause and effect relationships was acknowledged (See Rees *in current workshop proceedings*).

There will also be a need to define different thresholds for particular EcoQOs that reflect different stages in the recovery process, and for accounting for the degree of change in relation to distance away from the extraction activity i.e. what is viewed as acceptable at the extraction site may not be considered appropriate 5km away.

There was a general recognition that survey design was critical to the reliable detection of change in a selected measure of ecological status, especially so if EcoQOs took on legal significance.

The working group concentrated on the evaluation of metrics that may be employed to quantify changes in marine benthic communities, as these were considered most appropriate for the detection of localised ('near-field') impacts of aggregate extraction. However, it was recognised that the application of the EcoQ/EcoQO concept to fish communities, and its role in wider-area evaluations of ecological status, will be an important component of future investigations.

Selection of metrics

The working group sought to identify and rank metrics as a precursor to more detailed examination of their potential for application and formal EcoQOs. These were reviewed in terms of their relevance, applicability and current state of development with respect to the following two targets: national/regional biotope distribution and the protection of ecological functioning. The outcome is shown in Table 3. Initially the working group sought to identify suitable measures e.g. trophic indices. This list includes a range of techniques including conventional measures e.g. Shannon diversity index (see for example, Clarke and Warwick, 1994) as well as more recently proposed measures e.g. index of taxonomic distinctness (Warwick and Clarke, 2001). Each measure was then considered in terms of their application in coarse ground habitats (i.e. those habitats that are most at risk from the effects of aggregate extraction activities) and also their cost-effectiveness for quantifying effects. Issues raised, during the Working Group discussions, on the utility of each of the measures are also presented together with an indication of where further developments are likely.

Summary of conclusions reached by the Biological Working Group:

1. Further work was clearly needed before numerical limits can be assigned to each of the targets in Table 3.
2. The importance of adequate baseline (i.e. pre-dredging) data was acknowledged. Without such information it can be difficult to attribute cause and effect relationships (other than in respect of gross effects within the licensed area) and hence to quantify the spatial scale of effects.
3. Further research is required to facilitate cost-effective mapping of the spatial extent and integrity of biotopes, since this may have value both in determining 'baseline' (pre-dredging) status, and in subsequent monitoring of changes during and after the event of aggregate dredging.
4. The development and local application of ecosystem models is an important area for further research, since these may provide new insights into the nature of biological

responses to man-induced change at a more holistic level than at present. They may also facilitate the selection of key indicators for use in an EcoQO context.

5. The Benthic Response Index (see Smith *et al.*, 2001 and Pearson: current workshop proceedings) shows promise and needs further validation, especially in areas of coarse sediments. Other novel statistical indices such as the taxonomic diversity and taxonomic distinctness indices developed by PML, also have desirable attributes in an indicator context and merit further testing for their utility in discriminating changes associated with marine aggregate extraction.

6. There is a requirement for the development of conceptual models to accurately describe and quantify the effects on benthic communities of marine aggregate extraction (see Boyd: current workshop proceedings). These would provide an essential underpinning to the identification of appropriate EcoQOs, and also to the determination of associated thresholds for acceptable/unacceptable levels of change. The latter would need to convey ecological significance, in order to be useful in environmental management.

7. Presently, the scope for deriving and validating such models is limited by the availability of relevant data, but this should improve as more effort is directed at the assessment of environmental impacts. In particular, there is a need for further data on the time-scales for recovery of benthic populations following the cessation of dredging: the outcome of current research by CEFAS and others will be useful in this respect. In addition, further effort is required (using techniques of appropriate sensitivity) in order to delineate the spatial extent of dredging effects, both in physical and biological terms.

8. The value of consistent time-series information at impacted and reference stations was highlighted. Such information was essential for retrospective testing of the utility and robustness of potential EcoQOs, prior to their application in routine environmental management.

9. There was discussion on the utility of the 'mixing zone' concept, which was evolved in relation to pipeline discharges, and which was a necessary consideration in defining the spatial scale over which an activity might be in breach of environmental quality standards, in the immediate vicinity of discharge. The concept appears to be appropriate for consideration in relation to the immediate and localised consequences of passage of a draghead over the seabed during marine aggregate extraction.

References

ALLEN, J.H., 2001. Derivation of numerical predictive models from the NMP Dataset. Draft report to Scotland and Northern Ireland Forum for Environmental Research, Institute of Estuarine and Coastal Studies, University of Hull, Report: 421-Z100-D-2001, 68pp.

CODLING, I.D. AND ASHLEY, S.J., 1992. Development of a biotic index for the assessment of pollution status of marine and benthic communities. WRc plc Final Report NR 3102/1, 78pp.

CRISP, D.J., 1984. Energy flow measurements. In: Methods for the study of Marine Benthos pp.284-386. Holme, N.A. and McIntyre, A.D. (eds). 2nd Edition Oxford, Blackwell 387pp.

CLARKE, K.R. AND WARWICK, R.M., 1994. Change in Marine Communities: An approach to statistical analysis and interpretation. Plymouth, Plymouth Marine Laboratories, UK. 144pp.

DALSGAARD, J., JARRE-TEICHMANN, A., WALTERS, C AND PAULY, D., 1998. An approach to the modelling of persistent pollutants in marine ecosystems. ICES CM 1998/V:10.

HISCOCK, K. (EDITOR), 1998. Benthic marine ecosystems of Great Britain and the north-east Atlantic . Marine Nature Conservation Review. Peterborough. 404pp.

LAMBSHEAD, P.J.D, PLATT, H.M. AND SHAW, K.M., 1983. The detection of differences among assemblages of marine benthic species based on an assessment of dominance and diversity. J. Nat. History *17*:859-874.

MAFF, 1993. Analysis and interpretation of benthic community data at sewage sludge disposal sites. Aquatic Environment Monitoring Report. Number 37. Lowestoft 80pp.

SCHWINGHAMMER P., 1981. Characteristic size distributions of integral benthic communities. Can J. Fish Aquat. Sci. *38*:1255-1263.

SMITH, R.W., BERGEN, M., WEISBERG, B., CAIDEN, D., DALKEY, A., MONTAGNE, D. STULL, J.K., AND R.G. VELARDE, 2001. Benthic Response Index for assessing infaunal communities on the Southern California Mainland Shelf. Ecological Applications *11*(4): 1073-1087.

WARWICK, R.M AND CLARKE, K.R., 1993. Increased variability as a symptom of stress in marine communities. J. Exp. Mar. Biol. Ecol. *172*:215-226.

WARWICK, R.M AND CLARKE, K.R., 2001. Practical measures of marine biodiversity based on relatedness of species. Oceanogr. Mar. Biol. Ann. Rev. *39*:207-231.

Table 3. Key issues identified in Working Group 2

PRIMARY CONCERN	TARGET	FORMULATION OF MEASURE	CRITERION	STATE OF DEVELOPMENT OF QUALITY MEASURE			QUALIFYING REMARKS
				NOW	SOON	LATER	
national/regional biotope distribution	relevant biotopes	nature; size of biotope(s)	Protection of x% of the habitat/biotope from the effects of human activities		X		Dependent on the capacity to define biotopes. Acoustic techniques may have potential in this respect. Need for further information in relation to nature and distribution of offshore habitats.
protection of functioning	productivity	benthic production (Crisp, 1984)	Absolute changes can be viewed for their acceptability. Deviation from status quo	X			The significance of changes to e.g. demersal fish will be dependent on scale of dredging effects. Cost effectiveness is an important consideration.
	trophic links	functional group/trophic indices (Codling *et al*.,)	Deviation from status quo/local reference		X		Information on feeding habits not available for for all species; ITI index requires adaptation for coarse ground benthos, along with an underpinning 'effects' model for interpretation of results.
	biodiversity	multivariate techniques (e.g. Clarke and Warwick, 1994)	Deviation from status quo/local reference	X			Problems with communication of output to environmental managers, but these tools are essential for scientific interpretation. For univariate derivation of multivariate output see Warwick and Clarke, 1993.
		Benthic Response Index (e.g. Smith *et al*., 2001)	Deviation from status quo/local reference		X		Index is already formulated but needs testing/refinement in coarse ground habitats
		conventional diversity indices inc. graphical techniques (e.g. Lambshead *et al*., 1983; Clarke and Warwick, 1994)	Change relative to (in most cases) a local reference	X			Some further effort required to better understand changes on hard ground.
		taxonomic diversity (e.g. Warwick and Clarke, 2001)	The reference can be a globally derived measure	X			Some further effort required to better understand changes on hard ground.
		biotope definition (e.g. Hiscock, 1998)	Absolute changes can be viewed for their acceptability. Deviation from status quo	X			Biotope complexes more useful than individual biotope characterisation for the management purposes.
		primary variables (A,T,B) (e.g. MAFF, 1993)	Change relative to (in most cases) a local reference	X			Some question over reliability/cost-effectiveness of biomass measurements.
		size spectra (Schwinghammer,1981)	Change relative to (in most cases) a local reference		X (for certain components incl. macrofauna)	X (for entire benthic biota)	Likely to be an expensive option for entire benthic fauna; advances using image analysis techniques are likely. Sampling problems may arise for certain components (e.g. meiofauna)
		presence of indicator species/ keystone species	Absolute changes can be viewed for their acceptability.	X (keystone species)	X (indicator species)		More work needed to establish indicators of the effects of aggregate extraction;
		ecosystem' modelling e.g. ECOPATH; COASTPAC etc (ICES, Allen)	the ref is provided by predictive output -COASTPAC		X		Further developmental work needed; more relevant data on the effects of aggregate extraction is also required

WG 3. Define metrics appropriate for use as environmental indicators, and their state of development, practicality and feasibility:
Sub-group: Physical.

Chairman - Ceri James, BGS
Rapporteur – Jon Rees, CEFAS

Physical features which include parameters such as the geology, sediment and morphology of the seabed greatly influence the distribution and range of species and biological communities. It is these physical parameters which are disturbed during the marine aggregate extraction process. In terms of defining metrics, physical parameters are generally more readily mapped and quantified than biological parameters, especially through the use of modern geophysical techniques such as seismic reflection, side-scan sonar, AGDS (acoustic ground discrimination systems) and swath bathymetry. Grab and core sampling also provides physical data, which can be assessed and quantified.

The value of using physical parameters as metrics is enhanced by the fact that data is available on a regional scale in the UK marine environment and also the aggregate industry as a routine during aggregate resource surveys collects such data using geophysical and physical methods. These could be utilised as baseline data in constructing EcoQOs if the spatial density and timeliness of the data are appropriate. The aggregate industry also collects monitoring data during the dredging process that could feed into the development of physical metrics

The group began by discussing primary measurable parameters for the physical environment. They concluded by defining four primary parameters

1. Sediment;
2. Hydrodynamics;
3. Bathymetry/Morphology; and
4. Dredging method.

Detailed discussion then followed on the variables within these parameters that are important metrics of the status of the physical environment and would need to be considered when conducting impact assessments.

1. Sediment:

- grain size;
- depth/bedding;
- variability;
- density (void space);
- bedforms;
- porosity;
- permeability;

- regional distribution of sediments;
- biogenic debris;
- transport pathways;
- suspended sediment redeposition; and
- stability (critical erosion bed shear stress), consolidation, conditioning.

2. *Hydrodynamics:*
- waves (wave statistics, direction, return periods);
- boundary layer dynamics;
- tides (spring and neap max velocities, size of tidal ellipse);
- currents (residual);
- stratification;
- water column processes – density fronts/ROFI; and
- transport pathways.

3. *Bathymetry/Morphology:*
- changes in coastline e.g. beach height/character.
- depth of extraction at licence area
- changes in sand bank height and width
- changes in sand wave and megaripple height and distribution

4. *Dredging methods:*
- trailer suction dredging;
- anchor dredging;
- rates of production;
- loss of sediment/returns;
- dredge tracks;
- temporal/spatial history and intensity; and
- future/present management plans.

Not all these variables would be suitable for developing into practical metrics. Many may be constrained by local or regional controlling mechanisms, which may be difficult to measure or define. Similarly, variations in temporal scales may be important influences on physical parameters. These may include tidal and spring-neap cycles, seasonal winter-summer cycles, inter-annual and decadal cycles such as the NAO. Their significance as readily defined metrics is probably questionable

The group felt that if change in the extraction area was acceptable could it be defined as a metric in terms of a sediment grain size parameter? For example, a move in the Folk G-S-M triangle from one region to another. However, this requires prior knowledge of the ambient environment and its features, such as, banks, sheets and channels. The group also discussed the value of physical metrics in terms of making them understandable and meaningful to the general public. In this context would the change from say a 70/30 gravel/sand mix in an extraction area to a 40/60 mix after dredging be relevant to the public. If such measures are to be useful in public debate they must be underpinned by a clear explanation as to their ecological relevance.

After much discussion, it was suggested that the intermediate zone, which lies outside the aggregate extraction area, was probably the most important area to consider in terms of physical impact. This links into EcoQo 4 developed by Working Group 1 regarding secondary impact zones.

There was additional discussion about the ecological relevance of the physical parameters that were identified and whether they could be prioritised. Would it be possible to define a single physical metric for diverse habitats? And how do we numerically differentiate between say a gravely-sand and sandy-gravel?

The 1986 ICES North Sea Benthos survey (Kunitzer *et al*. 1992) identified 6 variables which were influential in determining the distribution of benthic communities – depth and sediment type being amongst the most important. Based on these findings, during the second day of the workshop a list of baseline data requirements was developed and the relative importance of each variable identified (Table 4). Those that were fundamental in defining a metric were given the value 1 on a two-stage scale with 2 confined to variables that were site specific in their value and/or required an analysis of their value in the context of the metric. This two-stage scale was also used for Table 5.

Table 4. Baseline data required

Variable	**Priority (1= fundamental, compulsory; 2 = site specific, expert judgement required)**
Bathymetry/morphology	1
Tides/waves/currents - review	1
Tides/waves/currents - detailed	2
Sub bottom geology	1
Water column dynamics	2
Background suspended load (nearbed)	2
Shoreline surveys	2

The utility of these variables and sampling techniques and their usefulness for operational monitoring was discussed. A second table was constructed with the general aim of obtaining more information from existing datasets (Table 5).

The group felt that any existing datasets utilised should fulfil a number of criteria including

- the data should be fit for the intended purpose;
- comprehensive survey design;
- adequate spatial and temporal coverage;
- sufficient data density; and
- follow defined sampling and analytical protocols.

Table 5. Operational monitoring parameters

Variable	Target Measure	Utility	Info Required
Bathymetry (depth)	• Depth of extraction	1	Soundings/Swath etc
Rate of extraction	• Volume (rate/frequency)	2	Production rates from vessel records
Bedforms/mobility	• Difficult to quantify	1	Sidescan, Video, Swath
Seabed Sediments (surface PSA)	• Gross change - acceptable limits of change	1	PSA of surface sediments Agreed protocols and sampling density
Dredging intensity + coverage	• % of new ground disturbed each year (Could relate to a cap on area being dredged)	1	EMS, zoning plans, long-term plans
Tides, currents , waves (including water column dynamics)	• Current velocities, wave heights (For confirmation of mathematical model outputs. Complements other variables)	2	Speed, direction, height etc
Sub-bottom geology	• Post dredging sub-bottom to agreed specification • Resource assessment pre, during and post dredging	1 1	Seismics, cores etc
Suspended sediment	• Statement of intent • Demonstrate compliance on special cases	2	*In situ* measurements using ADCP, Profilers and Landers. Also, input data from vessel records
Shoreline effects	• Beach sediment profile/volume	2	Coastal Impact study with beach profiles

Summary conclusions of the physical working group:

1) Baseline physical data is fundamental to the assessment of impacts of marine aggregate extraction,
2) Near and far field effects need to be considered within a regional perspective;
3) We must build on data already available, including that gathered for aggregate resource assessment and EIAs;
4) However, this requires data to be shared and made widely available to enable effective management of marine aggregate extraction;
5) A metadata database of data held by industry, consultants, government agencies etc. would an import first step towards achieving the previous point;
6) A GIS interface to available data would be very useful, particularly if accessible over the Internet; and
7) Biological and geological datasets need to be integrated to assist a better understanding of the impacts of marine aggregate extraction on the marine environment.

References

KUNITZER, A., BASFORD, D., CRAEYMEERSCH, J.A., DEWARUMEZ, J.M., DORJES, J., DUINEVELD, G.C.A., ELEFTHERIOU, A., HEIP, C., HERMAN, P., KINGSTON, P., NIERMANN, U,. RACHOR, E., RUMOHR, H. AND DE WILDE, P.A.J., 1992. The benthic infauna of the North Sea: species distribution and assemblages. ICES Journal of Marine Science *49*: 127-143.

4. CONCLUSIONS

The final afternoon of the workshop was a plenary session for all delegates at which the outcomes from the 3 working groups were presented and discussed. The following conclusions were derived from that discussion as well as the papers presented earlier and the discussions on them:

1. It was accepted that the principal environmental concerns with the activity of marine aggregate extraction arise from physical impacts at the seabed and the consequential biological responses. Chemical contamination is unlikely to be an issue in most cases due to the very low organic and clay content of commercial aggregate deposits and to the fact that most of these geological deposits would not have been exposed at the surface of the seabed prior to dredging.
2. The workshop agreed that it was not practical or feasible to develop specific EcoQOs for marine aggregate extraction in isolation from EcoQOs for other human activities impacting the seabed.
3. The workshop agreed that a set of overarching objectives was a necessary framework for the practical application of the EcoQO concept within which more specific metrics would operate to give more detailed information about the impacts of marine aggregate dredging. A number of those metrics would also be likely to be relevant to seabed disturbance by other human activities.
4. The workshop agreed that the 4 EcoQOs developed by Working Group 1 were appropriate as overarching objectives specific to the management of marine aggregate extraction.
5. It was agreed that the first 2 EcoQOs developed by Working Group 1 had general applicability to the impacts of human activities on the seabed.
6. It was agreed that objectives 3 and 4 developed by Working Group 1 could be the basis for the development of objectives for other human activities that impact the seabed.
7. EcoQOs which are in the future developed for a variety of human activities may overlap in their areas of application, for example in the case of a coastal locality which is subject to multiple uses. In such circumstances, the potential for conflicting and inconsistent EcoQOs must, at an early stage, be addressed by appropriate co-ordination of groups responsible for their development.
8. Working Groups 2 and 3 developed lists of metrics appropriate for use in assessing potential environmental impacts of marine aggregate extraction and reviewed their state of development, practicality and feasibility. The workshop agreed that these metrics were relevant to such assessments and to contribute to the development of overarching EcoQOs.
9. It was agreed that the Benthic Response Index (Working Group 2 report) shows particular promise as a metric but needs further validation, particularly for coarse sediments.

10. It was emphasised that adequate physical and biological baseline data was a necessity if cause and effect relationships were to be unequivocally attributed.

11. It was also emphasised that consistent time-series information at impacted and reference stations was essential for retrospective testing of the utility and robustness of potential EcoQOs, prior to their application in routine environmental management.

12. It was agreed that the sharing of data available to consultants, dredging companies, government agencies etc. would assist in a better understanding of the impacts of marine aggregate extraction and in particular, the integration of biological and geological datasets.

13. It was agreed that a number of the metrics developed by Working Groups 2 and 3 would be applicable to other human activities that disturb the seabed.

14. There was a need to develop conceptual models to accurately describe and quantify the effects on the seabed of marine aggregate extraction. They would provide an essential underpinning to the identification of appropriate EcoQOs, and also to the determination of associated thresholds for acceptable/unacceptable levels of change. The latter would need to convey ecological significance, in order to be useful in environmental management.

15. Presently, the scope for deriving and validating such models is limited by the availability of relevant data, but this should improve as more effort is directed at the assessment of environmental impacts. In particular, there is a need for further data on the time-scales for recovery of benthic populations following the cessation of dredging. In addition, further effort is required in order to delineate the spatial extent of dredging effects, both in physical and biological terms.

16. The development and local application of ecosystem models is an important area for further research, since these may provide new insights into the nature of biological responses to man-induced change at a more holistic level than at present. They may also facilitate the selection of key indicators for use in an EcoQO context.

17. The key research requirement identified was the facilitation of cost-effective mapping of the spatial extent and integrity of seabed biotopes. This will enable the identification of those considered to be the most important in terms of ecosystem structure and function, and therefore most in need of protection. This will also have value in determining 'baseline' (pre-dredging) status, and in subsequent monitoring of changes during and after the event of marine aggregate dredging. Such mapping will also be highly relevant to other human activities.

18. Specification of the overarching EcoQOs, the metrics to support them and the monitoring programmes to provide information about the status and trends in the EcoQOs and metrics will require further development. However, data exist for a number of variables presently targeted in a monitoring context, so that reference levels and degrees of acceptable change can be derived and then tested for their utility at least for some of the metrics discussed.

5. APPENDICES

APPENDIX 1. PARTICIPANTS

1. CEFAS

Chris Vivian	Burnham Laboratory, Remembrance Avenue, Burnham-on-Crouch, Essex, CM0 8HA. Tel: 01621 787253, Fax: 01621 784989. Email: C.M.G.Vivian@cefas.co.uk
Johanna Johnson	Lowestoft Laboratory, Pakefield Road, Lowestoft, Suffolk, NR33 08T. Tel: 01502 524540, Fax: 01502 513865. Email: J.Johnson@cefas.co.uk
Hubert Rees	Burnham Laboratory, Remembrance Avenue, Burnham-on-Crouch, Essex, CM0 8HA. Tel: 01621 787242, Fax: 01621 784989. Email: H.L.Rees@cefas.co.uk
Sian Boyd	Burnham Laboratory, Remembrance Avenue, Burnham-on-Crouch, Essex, CM0 8HA. Tel: 01621 787245, Fax: 01621 784989. Email: S.E.Boyd@cefas.co.uk
Audrey Jones	Burnham Laboratory, Remembrance Avenue, Burnham-on-Crouch, Essex, CM0 8HA. Tel: 01621 787361, Fax: 01621 784989. Email: A.Jones@cefas.co.uk
Jacqueline Jones	Burnham Laboratory, Remembrance Avenue, Burnham-on-Crouch, Essex, CM0 8HA. Tel: 01621 787293, Fax: 01621 784989. Email: J.Jones@cefas.co.uk
Andy Kenny	Burnham Laboratory, Remembrance Avenue, Burnham-on-Crouch, Essex, CM0 8HA. Tel: 01621 787258, Fax: 01621 784989. Email: A.J.Kenny@cefas.co.uk
Stuart Rogers	Lowestoft Laboratory, Pakefield Road, Lowestoft, Suffolk, NR33 08T. Tel: 01502 524267, Fax: 01502 513865. Email: S.I.Rogers@cefas.co.uk
Jon Rees	Lowestoft Laboratory, Pakefield Road, Lowestoft, Suffolk, NR33 08T. Tel: 01502 524383, Fax: 01502 513865. Email: J.M.Rees@cefas.co.uk

2. Government

Richard Emmerson	DEFRA, Marine and Waterways Division, 3/B8 Ashdown House, 123 Victoria Street, London, SW1E 6DE. Tel: 0207 944 5305. Email: Richard.Emmerson@defra.gsi.gov.uk
Margaret Wemys	DEFRA, Marine Land & Liability, Room 128, Nobel House, Smith Square, London SW1P 3HR. Tel: 0207 238 5715, Fax: 0207 238 5724. Email: Margaret.Wemys@defra.gsi.gov.uk
Tom Simpson	Department of Transport, Local Government and the Regions, Eland House, Zone 4/A2, Bressenden Place, London SW1E 5DW, Tel: 0207 944 3868, Fax: 0207 944 3859. Email: tom.simpson@dtlr.gsi.gov.uk
Tony Murray	Crown Estate Commission, 16 Carlton House Terrace, London SW1Y 5AH, Tel: 0207 210 4377, Fax: 0207 839 7847. Email: anthony.murray@crownestate.co.uk
Derek Moore	FRS Aberdeen Laboratory, PO Box 101, Victoria Rd, Aberdeen, AB11 9DB. Tel: 012 2429 5386, Fax: 012 2429 5511. Email: d.moore@marlab.ac.uk
Beth Greenaway	DEFRA, Cromwell House, Dean Stanley Street, London, SW1P 3JH. Email: Beth.Greenaway@defra.gsi.gov.uk

3. Agencies

Leigh Jones	English Nature, Northminster House, Peterborough, PE1 1UA. Email: leigh.jones@english-nature.org.uk
Nicola Rimington	Countryside Council for Wales, Plas Penrhos, Ffordd Penrhos, Bangor, Gwynedd , LL57 2LQ, Wales. Tel: 01248 385500. Fax: 01248 355782. Email: N.Rimington@ccw.gov.uk
Simon Gardner	Environment Agency, Existing EU & International Commitments, 2440 The Quadrant, Aztec West, Almondsbury, Bristol, BS32 4AQ. Tel: 01454-878850. Fax: 01454-878681. Email: simon.gardner@environment-agency.gov.uk
David Connor	Joint Nature Conservation Committee, Monkstone House, City Road, Peterborough, PE1 1JY. Tel: 01733 562626. Email: david.connor@jncc.gov.uk
Tim Mackie	Environment & Heritage Service, Calvert House, 23 Castle Place, Belfast BT 1 1FY, Tel: 02890 254 823, Fax: 02890 254865. Email: tim.mackie@irtu.detini.gov.uk

4. NGO

Lisa Browning (on behalf of the Wildlife Trusts & WWF)	c/o World Wildlife Fund UK, Panda House, Weyside Park, Cattleshall Lane, Godalming, Surrey, GU7 1XR.
Melissa Morton	Marine Conservation Society, 9 Gloucester Road, Ross-on-Wye, Herefordshire, HR9 5BU. Tel: 01989 566017, Fax: 01989 567815. Website: www.mcsuk.org.

5. Industry Representatives

Mark Russell	BMAPA, c/o 25A Astral Gardens, Hamble, Southampton, SO31 4RQ. Tel: 02380 458710, Fax: 02380 457707, mobile: 07870 596865. Email: russell@qpa.org. Website: www.bmapa.org
Andrew Bellamy	United Marine Dredging Limited (UMD), Resources Manager, UMD House, Shopwhyke Road, Chichester, West Sussex PO20 6AD, Tel: 01243 817200, Fax: 01243 817216. Email: agbellamy@umag.co.uk
Ian Selby	Hanson Aggregates Marine Limited, Senior Resources Manager, Head Office, Burnley Wharf, Marine Parade, Southampton SO14 5JF. Tel: 02380 828200, Fax: 02380 828248. Email: Ian.Selby@hanson-aggregates-m.com. Website: www.hanson-aggregates.com
Peter Hughes (ERM on behalf of Volker Dredging)	ERM (Environmental Resources Management), Eaton House, Wallbrook Court, North Hinksey Lane, Oxford OX2 0QS, Tel: 01865 384800. Email: post@ermuk.com

6. Others

Mike Elliott	Department of Biological Sciences, University of Hull, Hull, HU6 7RX. Tel: 01482 465 503, Fax: 01482 465 455, Email: M.Elliott@biosci.hull.ac.uk.
Paul Somerfield	Plymouth Marine Laboratory, Prospect Place, Plymouth, PL1 3DH. Tel: 01752 633449, Fax: 01752 633101. Email: PJSO@mail.pml.ac.uk
Ivor Rees	School of Ocean Sciences, University of Wales, Menai Bridge, Anglesey, North Wales, LL59 5EY. Tel: 01248 382 875, Fax: 01248 716 367. Email: oss058@bangor.ac.uk.
Chris Frid	Dove Marine Laboratory, Department of Marine Science & Coastal Management, University of Newcastle-upon-Tyne, Cullercoats, North Shields, NE40 4PZ. Tel: 0191 252 4850, Fax: 0191 252 1054. Email: c.l.j.frid@ncl.ac.uk.

Tom Pearson	Seas Ltd, c/o Dunstaffnage Marine Laboratory, Oban Argyll, PA34 4AD, UK. Tel: 01631 559000. Email: seas@dml.ac.uk
Michel Lemoine	IFREMER, Av. du Charles de Gaulle, BP 32, 14520 Port en Bessin, France. Tel: 02 31511300, Fax: 02 31511301. Email: Michel.Lemoine@ifremer.fr
Michel Desprez	GEMEL, 115 quai Jeanne d'Arc, 80230, Saint-Valéry sur Somme. Tel: 03 22268525, Fax: 03 22268774. Email: gemel.picardie@libertysurf.fr
Ceri James	British Geological Survey, Kingsley Dunham Centre, Keyworth, Nottingham, NG12 5GG. Tel: 011 5936 3467, Fax: 011 5936 3460. Email: jwcj@bgs.ac.uk
Keith Hiscock	Marine Biological Association of the UK, The Laboratory, Citadel Hill, Plymouth, PL1 2PB. Tel: 01752 633 335, Switch: 01752 633100, Fax: 01752 633 102. Email: khis@mba.ac.uk.
Mike Dearnaley	HR Wallingford, Howbery Park, Wallingford, Oxford OX10 8BA. Tel: 01491 835375. Email: mpd@hrwallingford.co.uk

APPENDIX 2. WORKSHOP PROGRAMME

11th October 2001

0900
Welcome and outline of the purpose of the workshop – Dr Chris Vivian, CEFAS.

0905
Regulation of marine aggregate extraction - Dr Tom Simpson, DTLR

0910
OSPAR's Ecosystem Approach to Management and EcoQ/EcoQOs – Dr Richard Emmerson, DEFRA

0930
Role of science in underpinning the development of summary measures of environmental impacts – Dr Tom Pearson, SEAS

1000
Introduction to Quality Objectives and Standards – Dr Mike Elliott, Hull University.

1030 MORNING TEA

1100
Potential applications of the EcoQO concept in relation to benthic impacts of marine aggregate extraction – Dr Hubert Rees and Dr Sian Boyd, CEFAS

1145
Feasibility and practicality of the EcoQOs concept in relation to fisheries impacts of marine aggregate extraction – Dr Stuart Rogers, CEFAS

1215
Update on current research on marine aggregate dredging impacts relevant to the workshop – CEFAS, BMAPA.

1245
Opportunity for questions and general discussion.

1300 – 1345 LUNCH

1345
Introduction to the Working Group topics and how the groups will operate – Dr Chris Vivian, CEFAS.

1400
Divide into Working Groups (WG) with all participants spending 20 minutes in each group:

WG1. Develop EcoQOs (analogous to EQOs) taking account practicality, feasibility, regulatory interests, conservation and fisheries interests.
[This group could also further explore (and seek to resolve) issues raised by inconsistent terminology, if there is time.]
Chair – Tom Simpson
Rapporteur – Johanna Johnson

WG2. Define measures appropriate for use as environmental indicators, and their state of development, practicality and feasibility:
Sub-group: Biological.
Chair – Hubert Rees
Rapporteur – Sian Boyd

WG3. Define measures appropriate for use as environmental indicators, and their state of development, practicality and feasibility:
Sub-group: Physical.
Chair – Ceri James
Rapporteur – Jon Rees

1500
Brief report back from each Working Group

1530 AFTERNOON TEA

1600
Divide into Working Groups*

1930
Workshop dinner at The Crooked Barn Restaurant

* Participants rotated through all the working groups with a short brainstorming session in each to flush out all the issues that needed to be considered. Subsequently, participants selected a particular working group to participate in on Day 2.

12th October 2001

0900
Resume discussions in working groups

1030 MORNING TEA

1100
Final working group session.

1200
General discussion of areas needing further research in order to develop EcoQOs or other measures of environmental impact for marine aggregate extraction

1230 – 1315 LUNCH

1315
Presentation of outcomes of the 3 working group's discussions by the Chair/Rapporteur of each working group and the summary of the discussion on future research needs (15 minutes each).

1415
Discussion of outcomes of working groups, agreement on future actions and identification of areas of further research.

1530 AFTERNOON TEA

1600
Closing address – Dr Chris Vivian.

APPENDIX 3. FURTHER READING

1. Barnett, V. and O'Hagen, A., 1997. Setting Environmental Standards: The statistical approach to handling uncertainty and variation. Chapman and Hall, London.

2. Benthos Task Team, 1996. Monitoring and Assessment of the Marine Benthos at UK Dredged Material Disposal Sites. Prepared by the Benthos Task Team for the Marine Pollution Monitoring Management Group, Group Co-ordinating Sea Disposal Monitoring Department of Agriculture and Fisheries for Scotland, Report Number 21, 35pp.

3. Brown, A.R. and Shillabeer, N., 1996. Development of a biologically based environmental quality standard from a long-term benthic monitoring programme in the North Sea, Oceanologica Acta, *20(1):* 275-282.

4. DETR, 2001. Development of Ecological Quality Objectives with Regard to Eutrophication, Department of the Environment, Transport and the Regions, Final Report, March 2001.

5. DETR, 2001. Draft Marine Minerals Guidance Note 2: Guidance on the Extraction by Dredging of Sand, Gravel and Other Minerals from the English Seabed. Consultation Paper Department of the Environment Transport and the Regions, London, 42pp.

6. Elliott, M., 1996. The Derivation and Value of Ecological Quality Standards and Objectives, Editorial Marine Pollution Bulletin *22(*11): 762-763.

7. Environment Agency, 2001. Tracers: Priorities for Environmental Quality Standard Development Research and Development Technical Report, Environment Agency.

8. EU, 2000. Common Position adopted by the Council of 22 October 1999 with a view to the adoption of a Directive establishing the framework for Community action in the field of water policy, 9085/3/99/Rev3, Annex V, DG.

9. Gilliland, P., 1998. The value and applications of EQO/EQS in nature conservation and environmentally sustainable development, English Nature.

10. Lanters, R.L.P., Skjoldal, H.R. and Noji. T.T., 1999. Ecological Quality Objectives for the North Sea. Basic Document for the Workshop on EcoQOs for the North Sea, Scheveningen, The Netherlands, RIKZ Report 99.015.

11. Marine Pollution Monitoring Management Group, Group Co-ordinating Sea Disposal Monitoring (MPMMG-GCSDM), 1997. Final Reports of the Metals Task Team and the Organics Task Team, Science Series, Aquatic Environmental Monitoring Report No. 49, 51 pp.

12. OSPAR, 1998. OSPAR Strategy for the Protection and Conservation of the Ecosystems and Biological Diversity of the Maritime Area, Ministerial Meeting of the OSPAR Commission 1998.

13. OSPAR, 1999. The Further Development of Ecological Quality Objectives within OSPAR, Issues and work plan agreed by IMPACT 1999. IMPACTS 99/15/1, Annex 5, 1999.

14. OSPAR, 1999. Strategy for the Further Development of Ecological Quality Objectives within OSPAR (Annex 5), Ministerial Meeting of the OSPAR Commission 1999.

15. OSPAR, 1999. Report from the Workshop on Ecological Quality Objectives (EcoQOs) for the North Sea and proposal for follow up workplan. IMPACT 99/3/1-E, May 1999.

16. RCEP, 1998. Setting Environmental Standards, Royal Commission on Environmental Pollution, Twenty-first Report, October 1998.

17. Rees, H.L., Moore, D.C., Pearson, T.H., Elliott, M., Service, M., Pomfret, J. and Johnson, D., 1990. Procedures for the Monitoring of Marine Benthic Communities at UK Sewage Disposal Sites, Department of Agriculture and Fisheries for Scotland, Report Number 18, 79 pp.

18. Skjoldal, H.R., 1999. Overview Report on Ecological Quality (EcoQ) and Ecological Quality Objectives (EcoQOs). Report prepared within the framework of the OSPAR Commission, Institute of Marine Research, June 1999.

19. Rees, H.L. and Pearson, T.H., 1992. An Approach to the Setting of Environmental Quality Standards at Marine Waste Disposal Sites, Marine Environmental Quality Committee, ICES CM 1992/E: 33.