Healthy & Biologically Diverse Seas Evidence Group Technical Report Series:

Evaluation and gap analysis of current and potential indicators for Sediment Habitats

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Preface

The UK Marine Monitoring and Assessment Strategy (UKMMAS) aims to provide coordinated and integrated marine monitoring programmes which support periodic assessments of the state of the UK marine environment. The strategy aims to provide vital data and information necessary to help assess progress towards achieving the UK's vision of clean, healthy, safe, productive and biologically diverse seas. The overarching strategy is supported and delivered by four evidence groups; Clean and Safe Seas Evidence Group (CSSEG); Productive Seas Evidence Group (PSEG); Healthy and Biologically Diverse Seas Evidence Group (HBDSEG) and Ocean Processes Evidence Group (OPEG). These groups are responsible for implementing monitoring and observations programmes to contribute to ecosystem-based assessments of marine environmental status.

As part of the HBDSEG programme of work, a series of reviews of environmental indicators was undertaken for the following marine ecosystem components:

1. Rock and biogenic reef habitats
2. Sediment habitats
3. Deep sea habitats
4. Seabirds and waterbirds
5. Cetaceans
6. Seals
7. Plankton
8. Microbes

The aim of the reviews was to evaluate a wide range of currently available and potential indicators for marine biodiversity monitoring and assessment. This task was undertaken particularly to inform future needs of the EU Marine Strategy Framework Directive (MSFD). The work was carried out by a group of consultants and contributors and was managed by JNCC.

Each review included a process to evaluate indicator effectiveness against a set of specified scientific and economic criteria. This process identified those indicators of activity, pressure, state change/impact and ecosystem structure and function that were considered to be scientifically robust and cost effective. The indicators which met these criteria were then assessed for inclusion within an overall indicator suite that the reviewers considered would collectively provide the best assessment of their ecosystem component's status. Within the review, authors also identified important gaps in indicator availability and suggested areas for future development in order to fill these gaps.

This report covers one of the ecosystem components listed above. It will be considered by HBDSEG, together with the other indicator reviews, in the further development of monitoring and assessment requirements under the MSFD and to meet other UK policy needs. Further steps in the process of identifying suitable indicators will be required to refine currently available indicators. Additional indicators may also need to be developed where significant gaps occur. Furthermore, as the framework within which these indicators will be used develops, there will be increasing focus and effort directed towards identifying those indicators which are able to address specific management objectives. There is no obligation for HBDSEG or UKMMAS to adopt any particular indicators at this stage, based on the content of this or any of the reports in this series.

This report has been through a scientific peer review and sign-off process by JNCC and HBDSEG. At this time it is considered to constitute a comprehensive review of a wide range of currently available and potential indicators for this marine ecosystem component.

Summary

The overall aim of this review was to identify the most effective indicators of marine ecosystem state, pressure and impacts to allow a scientifically robust assessment of marine environmental status. This chapter focuses on intertidal, coastal subtidal (to a depth of 50 m) and shelf (a depth of 50-200 m) sediments. It aims to present an assessment of the applicability of existing indicators, to identify where modifications might be appropriate and to identify significant gaps. As such, it includes indicators of sedimentary conditions in marine, coastal and estuarine areas and biological indicators based on benthic angiosperms, microscopic and macroscopic algae and invertebrates.

A literature review of existing indicators was carried out followed by evaluation and scoring according to a pre-determined set of scientific and economic criteria. This evaluation was carried out using a database created by JNCC. The outputs of this analysis were used to identify gaps in the monitoring where additional indicators may be required. In general, at least one (usually several) indicator of state/impact and/or pressure was identified for each pressure. In contrast, there were significant gaps for ecosystem functioning. Most of the indicators (with the exception of those relating to bioturbation) which related to ecosystem function were only indirect measures for primary and secondary production.

A total of 128 indicators were evaluated, relating to sediment quality, primary and derived indicators relating to the zoobenthos, indicators of the status of saltmarsh, seagrass and macroalage (opportunistic algae and fucoids), indicators of the status, spatial extent and distribution of key habitats and species (characterising, of conservation importance, indicator species and non-indigenous species) and a number of biomarkers. Of these 128 indicators, 115 were automatically recommended by the database output and 100 of these were accepted as valuable indicators, covering all the sedimentary ecosystem components mentioned above, for consideration in routine monitoring.

Contents

1 Introduction

In recent years a considerable amount of new environmental legislation has been presented with the aim of monitoring, conserving and protecting the marine environment (e.g. the Marine Strategy Framework Directive, etc). Indicators have increasingly become an important tool to monitor the impacts of human activities at sea, when combined with a clear policy framework describing the vision and goals for the ecosystem. Indicators are required to monitor the progress made towards meeting operational objectives and to guide policy makers and inform the public of the effectiveness of both legislation and management in improving the state of the environment.

In light of recently adopted and existing policy drivers (e.g. Marine and Coastal Access Bill, Charting Progress and OSPAR Quality Status Reports, and the Marine Strategy Framework Directive) there is an increasing emphasis on the need to identify indicators that are useful in meeting specific regulatory needs. Also there is a need to assess environmental status against higher level descriptors set out under the Marine Strategy Framework Directive (MSFD).

The present study is part of a larger initiative to scope most of the indicators used presently in national and international processes (e.g. SEBI, CBD etc) or currently published in the scientific literature, for the major components of the ecosystem. These will be assessed by the relevant Evidence Groups of the UK Marine Monitoring and Assessment Strategy (UKMMAS) in relation to a range of specific legislative drivers.

Indicators are defined as parameters or values derived from parameters that describe the state of the environment and its impact on humans, ecosystems and materials, the pressures on the environment, the driving forces and the responses steering that system. Indicators go through a process of selection and/or aggregation process to enable them to steer that action (EEA, 2007). This report addresses the process of selection, aggregation and assessment of indicators relevant to intertidal, coastal (down to 50 m) and shelf sediment (50-200 m) habitats.

1.1 Aims & objectives of report

The overall aim of this review is to identify the most effective indicators of marine ecosystem state, pressure and impacts to allow a scientifically robust assessment of marine environmental status. This chapter will focus on intertidal, coastal subtidal (to a depth of 50 m) and shelf (a depth of 50-200 m) sediments. It aims to present an assessment of the applicability of existing indicators, to identify where modifications might be appropriate and to identify significant gaps. As such, it includes indicators on sedimentary conditions in marine, coastal and estuarine areas and biological indicators based on benthic angiosperms, microscopic and macroscopic algae and invertebrates.

1.1.1 Objectives

a Review existing indicators for intertidal and subtidal sediment habitats.
b Evaluate the effectiveness of the indicators against scientific and economic criteria.
c Review the indicators against relevant pressures and important aspects of ecosystem structure and function.
d Identify significant gaps and identify indicators which may fill these gaps.
e Recommend a set of indicators (current and potential) for intertidal and subtidal sediment habitats that are scientifically and economically effective and could be used in an integrated monitoring and assessment programme.

1.2 Work undertaken in report

The present study was conducted by the Institute of Estuarine & Coastal Studies (IECS) and The Centre for Environment, Fisheries and Aquaculture Science (Cefas). A comprehensive review of the literature and the major policy drivers in the UK and Europe was undertaken to identify all indicators relevant to sediment habitat assessment. A wide range of colleagues (both within and outside Cefas and IECS) were contacted, especially those involved in Water Framework Directive Pressure Assessment for Coastal waters, CSEMP offshore monitoring, and OSPAR JAMP and EcoQO frameworks. A compilation of objectives by the Evidence Groups of the UKMMAS was used to identify policy drivers for which monitoring is undertaken.

This resulted in a structured appraisal describing the components to which the indicator applies (sediment physical structure, topography, biological assemblage structure or threatened or declining species abundance/biomass, etc), and its current status (in use or under development).

A critical review of indicators was conducted using objective review criteria (section 5.2). These were based on criteria for identifying desirable properties of indicators used in previous evaluations of the EcoQO framework by ICES. These were:

a Relatively easy to understand by non-scientists and those who will decide on their use;
b Sensitive to a manageable human activity;
c Relatively tightly linked in time to that activity;
d Easily and accurately measured, with a low error rate;
e Responsive primarily to a human activity, with low responsiveness to other causes of change;
f Measurable over a large proportion of the area to which the indicator is to apply;
g Based on an existing body or time series of data to allow a realistic setting of objectives.

Indicator details were entered into a database (designed and maintained by JNCC) and were automatically recommended or rejected based on the outcome of the scientific and economic evaluation. Recommended indicators were further evaluated against the relevant pressures and components of ecosystem functioning. Gaps where indicators were lacking were identified and a final suite of indicators, considered to be the most effective, was recommended.

1.3 Introduction to the ecosystem component of interest

The monitoring of sediment habitats and their associated benthic communities is necessary to determine the extent and health of the habitats and the integrity of the physical and biological features (Elliott *et al* 1998). This helps to determine natural variability and to identify departures from that due to anthropogenic impacts.

This report addresses indicators for all habitats and species of conservation importance within sediment environments in the intertidal, coastal subtidal (to a depth of 50 m) and shelf (a depth of 50-200 m) as listed in the Habitats Directive, the UK Biodiversity Action Plan (BAP) website and those given protection under the OSPAR convention. Habitats and marine landscapes include intertidal mud and sand flats, vegetated coastal shingle, coastal saltmarsh, coastal sand dunes, *Mytilus edulis* beds (OSPAR list), sheltered muddy gravels, reedbeds, saline lagoons, intertidal seagrass beds, gravels and cobble reefs. Soft sediment habitats within the subtidal environment include the important biotope complexes of current-swept sands, maerl beds, seagrass beds (*Zostera marina; Posidonia oceanic* and *Ruppia sp.)*; mud in deep water, sea pens and burrowing megafauna and sublittoral sands and gravels.

In the intertidal environment, mudflats and sandflats are a widespread habitat type throughout the UK and form a major component of estuaries, adjacent sedimentary coastal areas, embayments and semi-enclosed areas including lagoons (Davies *et al* 2001). As such they are amongst the most dominant marine and estuarine habitats and cover areas from a few hectares to several square kilometres within a site and several times this within any geographical area (Elliott *et al* 1998). Subtidal sedimentary habitats (down to 200m) cover large areas of the available continental shelf and thus are integral components of other designated biotope complexes.

It is considered that the importance of these habitats centres on their role in the biological and physical functioning of the ecosystems. For example, mudflats are highly productive and the invertebrate populations they support provide an important food source for predators, such as birds, fishes and mobile epibenthic invertebrates. Additionally, they play an important role in coastal protection. The protection of this functioning relies on maintaining the size of area, the tidal elevation and substratum type plus maintaining an input of colonising organisms and the predator populations (Elliott *et al* 1998). Ecosystem functioning depends on its structure, diversity and integrity. Alteration or disturbance of one or several components of marine ecosystems can have strong effects on higher or lower trophic levels, depending on whether food webs are controlled by resources or by predators.

1.4 Policy background

The Water Framework Directive requires that water bodies are classified in terms of the anthropogenic pressures to which they are subjected in order to identify those that can be described as ‘Heavily Modified Water bodies’ (HMWB). The HMWB are required only to have ‘Good Ecological Potential’ rather than to meet ‘Good Ecological Status’, i.e. their ecological status would be good were it not for the presence of physical/hydromorphological modifications (Borja & Elliott 2007). This approach requires the identification and description of significant anthropogenic pressures and their respective impacts on the physical and hydromorphological characteristics of the water body. Each activity can exert one or more pressures but it is important to identify the scale of each pressure in isolation and in combination with other pressures. Aubry & Elliott (2006) achieved this by developing a scoring system to represent the percentage of the seabed, which would be impacted by each pressure.

Indicators of sedimentary environmental quality include direct measurements of sediment properties (physical and chemical) together with characterisation of the biological communities. The major pressures causing impact have been divided (under the Marine Strategy Directive) as those causing the following:

- Physical loss, damage or disturbance;
- Interference with natural hydrological processes;
- Contamination by hazardous substances
- Nutrient and organic enrichment
- Biological disturbance
- Climate change

There is a need for a framework for the more effective co-ordination of marine monitoring in the UK as has been initiated through the development of the UKMMAS. This is in order to assess whether policies are meeting their objectives and to develop a strategic response to new monitoring requirements under OSPAR, EU and other auspices. An improved, co-ordinated reporting system based on indicators will form part of this framework and, properly constructed, will also have a pivotal role in delivering an ecosystem-based approach to environmental management.

Under the evolving EU Marine Strategy Framework Directive it will be necessary for Member States to achieve Good Environmental Status in their marine waters by 2020. Through the MSFD member states have adopted a set of specific qualitative standards, the so called 'Good Environmental Status' (GES) descriptors that are to be reported against from 2012. Key to this reporting process is the definition of what GES will practically mean, including relevant targets and indicators by 2012, and a monitoring programme to be established by 2014. Only a proportion of the ecosystem components and pressures identified in the Directive will be relevant to UK waters, and there is ongoing activity within OSPAR and ICES to identify a framework for identifying such priorities. When complete, this framework will allow key pressures and impacts in the marine environment to be identified, and suggest areas within which indicator development is most urgently required. The present study is part of a larger body of work that will scope available indicators for the major components of the ecosystem, and allow their subsequent assessment by the relevant Evidence Groups of the UKMMAS.

Until recently, water quality was at the forefront of estuarine and coastal management. The parameters measured were mostly related to human health, including:

- Physico-chemical water characteristics
- Toxicology
- Bacteriology
- Chlorophyll a

In Europe, the Water Framework Directive (WFD) (European Commission 2000) - and the recently proposed EU Marine Strategy Framework Directive – have established a framework for the protection of groundwater, inland surface waters, estuarine (transitional) waters and coastal waters and eventually out to the 200nm or mid line. As highlighted by Borja (2005), the WFD has several objectives: to prevent water ecosystem deterioration, to protect and to enhance the status of water resources but the most important aspect is to achieve a 'Good Ecological Status' (GES) for all waters, by 2015. In essence, the WFD requires a water body to be compared against a reference condition and then its ecological status designated - if the water body does not meet good or high ecological status, i.e. it is in moderate, poor or bad ecological status, then remedial measures have to be taken (e.g. source of pollution has to be removed).

The WFD ecological status is defined in relation to the health of five biological elements in coastal and transitional waters of which three are benthic (the benthic macrofauna, macroalgae and the angiosperms such as seagrasses and saltmarshes) - the others are phytoplankton and fishes (the latter is only assessed in transitional waters). The WFD centres on the influence of hydromorphology in affecting the biota although the chemical status of the water body is also assessed. The reference condition relates to what is expected for an area and is defined according to one of four ways:

- by choosing similar but unimpacted areas (i.e. a physical control similar to the test area but without human influences),
- by hind casting (i.e. assessing what the area was like at some previous time),
- by deriving predictive models (i.e. predicting the benthic community of an area based on the physical characteristics - see below) and lastly,
- according to the Directive, if all else fails then by using expert judgement. Similarly Descriptors for the MSD also exist and are presented in the box below.

Under the MSFD, descriptors of Good Environmental Status are as follows:

a Biological diversity is maintained. The quality and occurrence of habitats and the distribution and abundance of species are in line with prevailing physiographic, geographic and climatic conditions.
b Non-indigenous species introduced by human activities are at levels that do not adversely alter the ecosystems.
c Populations of all commercially exploited fish and shellfish are within safe biological limits, exhibiting a population age and size distribution that is indicative of a healthy stock.
d All elements of the marine food webs, to the extent that they are known, occur at normal abundance and diversity and levels capable of ensuring the long-term abundance of the species and the retention of their full reproductive capacity.
e Human-induced eutrophication is minimised, especially adverse effects thereof, such as losses in biodiversity, ecosystem degradation, harmful algae blooms and oxygen deficiency in bottom waters.
f Sea floor integrity is at a level that ensures that the structure and functions of the ecosystems are safeguarded and benthic ecosystems, in particular, are not adversely affected.
g Permanent alteration of hydrographical conditions does not adversely affect marine ecosystems.
h Concentrations of contaminants are at levels not giving rise to pollution effects.
i Contaminants in fish and other seafood for human consumption do not exceed levels established by Community legislation or other relevant standards.
j Properties and quantities of marine litter do not cause harm to the coastal and marine environment.

k Introduction of energy, including underwater noise, is at levels that do not adversely affect the marine environment.

To determine the characteristics of Good Environmental Status in a Marine Region or Sub-Region as provided for in Article 8(1), Member States should consider each one of the generic qualitative descriptors listed in the Annex to the Directive in order to identify those descriptors which are to be used to determine good environmental status for that Marine Region or Sub-Region. When a Member State considers that it is not appropriate to use one or several of those descriptors, it should provide the Commission with a suitable justification.

At this juncture the joint ICES / JRC Task Groups assigned to develop the criteria and methodological standards have started to define what the working definitions will be. Although this is still very much work in progress, some initial working principals are taking shape, e.g. the Biodiversity descriptor (1) is expected to cover all species groups and habitat types (including their communities) and in doing so describe the structural aspects of ecosystem components; the food web descriptor (4) will consider the process aspects of marine food webs, especially the rates and directions of energy transfer; and the sea floor integrity descriptor (6) will consider the functional aspects of biotic and abiotic ecosystem components.

1.5 OSPAR/UKMMAS assessment framework background

The assessment framework developed by JNCC was first presented to the OSPAR Convention's Biodiversity Committee in February 2007 and has since gained wide support across OSPAR as a tool to guide the development of a strategic approach to biodiversity monitoring. It has been particularly welcomed for its potential benefit in meeting the needs of the Marine Strategy Framework Directive (MSFD).

The framework takes the form of a matrix which relates ecosystem components (e.g. deep-seabed habitats) to the main pressures acting upon them (e.g. physical disturbance to the seabed). The ecosystem components have been correlated with components used by OSPAR and the MSFD. The columns of the matrix are a generic set of pressures on the marine environment, which are based on those used by OSPAR, MSFD and the Water Framework Directive (WFD). A 3-point scale of impact (low, moderate, high) reflects the degree of impact each pressure has on an ecosystem component. Each cell of the matrix has additionally been populated with a set of known indicators[1], derived from statutory and non-statutory sources, which are used to monitor and assess the state of that ecosystem component. The assessment matrix helps to highlight priorities for indicator development and monitoring programmes, based on the likely degree of each impact on the ecosystem component in question.

Since 2007 this approach has also been introduced to the UK's Marine Monitoring and Assessment Strategy (UKMMAS) and is being further developed by the Healthy and Biologically Diverse Seas Evidence Group (HBDSEG). The intention has been to have parallel development at UK and OSPAR levels which will help ensure similar biodiversity

[1] Note: cells of the matrix where impacts have been identified currently contain a number of species and habitats on protected lists (OSPAR, Habitats Directive), which could potentially be used as indicators of the wider status of the ecosystem component which they are listed against. Should this be appropriate, certain aspect of the species or habitat (e.g. its range, extent or condition) would need to be identified to monitor/assess.

strategies are developed at national and international levels. It is also envisaged that the development process will benefit from wide input across OSPAR Contracting Parties.

The overall goal of the UKMMAS is to implement a single monitoring framework that meets all national and international multiple policy commitments (UKMMAS, 2007). This will identify if there are any significant gaps in the current monitoring effort and aim to minimise costs by consolidating monitoring programmes. To help meet this goal, the assessment matrix has been developed with HBDSEG to provide a useful framework that analyses components of an ecosystem and their relationships to anthropogenic pressures. The framework aims to encompass three key issues: an assessment of the state of the ecosystem and how it is changing over space and time, an assessment of the anthropogenic pressures on the ecosystem and how they are changing over space and time, and an assessment of the management and regulatory mechanisms established to deal with the impacts.

The further development of the assessment framework has been divided into five shorter work packages: 1) assessment of pressures, 2) mapping existing indicators to the framework, 3) review of indicators and identification of gaps, 4) modifying or developing indicators and 5) review of current monitoring programmes. The following work will contribute to work package 3 and will critically review indicators, identify gaps and recommend an overall suite of the most effective indicators for the ecosystem component in question.

1.6 Definitions used within the report and analysis:

Definitions of activity, pressure, state change/ecological impact and ecosystem structure and function are used within this report as follows (adapted from the 2008 CP2 methodology[2]):

***Activity* –** Human social or economic actions or endeavours that may have an effect on the marine environment e.g. fishing, energy production.

***Pressure* -** the mechanism (physical, chemical or biological) through which an activity has an effect on any part of the ecosystem e.g. physical disturbance to the seabed.

***State change/ecological impact* –** physical, chemical or biological condition change at any level of organisation within the system. This change may be due to natural variability or occurs as a consequence of a human pressure e.g. benthic invertebrate mortality.

***Ecosystem structure and function* –** ecosystem level aspects of the marine environment (i.e. structural properties, functional processes or functional surrogate aspects) which are measured to detect change at higher levels of organisation within the system (i.e. changes at ecosystem scales), that is not attributable to any pressure or impact from human activity e.g. natural changes in species' population sizes. Please see Annex 4.

Defined pressures list:

[2] Robinson, L.A., Rogers, S., & Frid, C.L.J. 2008. *A marine assessment and monitoring framework for application by UKMMAS and OSPAR – Assessment of Pressures and impacts* (Contract No: C-08-0007-0027 for the Joint Nature Conservation Committee). University of Liverpool, Liverpool and Centre for the Environment, Fisheries and Aquaculture Science, Lowestoft.

The standard list of pressures against which indicators for this ecosystem component are reviewed is taken from the generic pressures list in the latest version (v11) of the UKMMAS / OSPAR assessment framework. Those pressures which are relevant to the ecosystem component (i.e. those that cause any impact on it) are used within the critical indicators review, gap analysis and this report.

2 Methods and data sources

An extensive literature review was carried out in order to identify indicators which are currently in use in Europe and those which are currently under development or proposed for use in the North East Atlantic region. Sources of information included:

- Scientific literature (sourced using Web of Science; Scopus);
- The UKMMAS marine protocols database (http://www.wrcplc.co.uk/marineprotocols/, created and maintained by WRc Plc and IECS);
- Publications by Defra and the Scottish Government such as Charting Progress (http://www.defra.gov.uk/marine/science/monitoring/stateofsea.htm) and *Scotland's biodiversity indicators* http://www.scotland.gov.uk/Publications/2007/10/08091435/46);
- Davies *et al* (2001), Marine Monitoring Handbook and Common Standards Monitoring Guidance;
- WFD United Kingdom Technical Advisory Group (UK TAG) documents;
- European Environment Agency;
- Langenberg & Troost (2008);
- Direct discussion with organisations responsible for monitoring activities (e.g. Conservation Agencies, Cefas, Environment Agency);
- Results from European research programmes including BIOMARE (Implementation and networking of large-scale long-term Marine Biodiversity research in Europe) and BEEP (Biological Effects of Environmental Pollution in marine ecosystems).

The literature review enabled identification of the relevant indicators and an initial assessment of their relative merits and drawbacks, based on their application and performance in a research and, where information was available, monitoring context. This provided the basis for the scientific and economic evaluation described in section 5.2. Following the literature review, details of individual indicators were entered into a database, designed by JNCC, including a description of the indicator and the geographical extent of its use, the ecosystem components to which it can be applied, the relevant human activities and pressures and a scientific and economic evaluation of the indicator. Scientific evaluation included an assessment of the sensitivity, accuracy, specificity, performance, simplicity, responsiveness, spatial applicability, relevance to management, validity and ease of communication to non-scientists. Economic criteria included the platform requirement for surveys (e.g. ship time vs survey on foot), equipment requirements and staff time involved in sample collection, processing, analysis and quality assurance, hence being focussed as value for money and cost-effectiveness. Each indicator will be assessed against these criteria as detailed in section 5.2 to give overall scores of good, moderate or poor for each indicator. Based upon this assessment, indicators were either recommended or rejected automatically. These recommendations were then individually assessed and the decision accepted or rejected, with justification.

In order to avoid double counting and artificially inflating the potential cost of monitoring, indicators were classified as 'direct' or 'derived'. That is, where a parameter is used in the calculation of an index. Additionally, indicators were not aggregated at a high level (although they have been categorised in this way for reporting purposes) within the database to ensure that the scientific and economic evaluation were carried out for each individual parameter. For example, indicators relating to habitat condition were entered as separate parameters and the relevant habitats/ecosystem components were identified. Exceptionally, indicators of sediment quality were aggregated into overall chemical groups (metals, PAH compounds etc) since the analysis of one sample will generate a large list of determinands. To enter each individual compound would give a false indication of the cost of including these indicators.

The database output, following indicator evaluation, was a matrix presenting each indicator against the relevant pressures and aspects of ecosystem structure and function. This matrix was used to carry out a gap analysis, to make final recommendations of the best suite of indicators for inclusion in monitoring and to identify further research needs.

3 Review of the existing indicators and critical evaluation

3.1 Current indicators summary

At the highest level of aggregation, indicators relevant to sediment habitats can be divided into seven categories. Within each, several indicators exist which may be relevant to different habitats or ecosystem components and which may be composed of different parameters. For example, the list of Scottish Biodiversity Indicators includes 17 state indicator categories, five of which are related to the marine environment: BAP priority species, BAP priority habitats, notified species/habitats in favourable condition and invasive non-native species (Scottish Government, 2007). The European Environment Agency (EEA, 2007) detail a much larger number of high level indicator classes including various physico-chemical features of the environment, habitat types (including special habitats), biological elements including characteristics of the bottom fauna, introduced or invasive species and aquatic flora (angiosperms and macroalgae), nutrient status and chemical contamination. Langenberg & Troost (2008) identified 199 indicators belonging to 8 different frameworks (policy drivers or tools developed for the application of indicators in environmental monitoring). This section will follow a similar approach giving specific indicator and parameter details under each section.

3.1.1 Indicators of sediment quality (physical and chemical)

Indicators of sediment quality include parameters such as grain size distribution, sediment composition, organic content, redox potential, deposition/erosion characteristics, topography and bathymetry and any pollutants that might be stored in the sediment itself or pore water. These parameters provide an overall indication of habitat quality, any change to which would directly impact upon the benthic and epibenthic communities present. The physical properties are strongly related to biological community structure and, as such, can be considered important and effective measures of environmental state or change. Additionally, there are various simple, inexpensive but effective ways of characterising sediments making sediment analysis (particularly in terms of grain size distribution and organic content) not only a widely used but essential indicator which can be applied to subtidal sedimentary environments.

It should be noted that changes in the physical properties of sediment, be it a change in particle size distribution or removal or deposition of sediment, can be caused by both natural hydrological processes, particularly in dynamic environments, and a variety of anthropogenic activities. It is often difficult to make a distinction between the potential causes of change. For example, extraction of non-living resources (quarrying, gravel/sand, dredging and oils and gas exploration), beach replenishment, disposal of solid waste, mariculture, extraction of living resources (fisheries related activities such as benthic and hydraulic dredging), shipping, and construction work all cause one or more forms of habitat damage in terms of change to siltation rate, abrasion or habitat removal, habitat loss, in terms of change in substratum type, and smothering or sealing. Therefore, sediment properties as indicators should not be used in isolation. Effectiveness can be increased by measuring these parameters and interpreting them in line with the activities taking place and at what scale.

Changes in the physical structure of sediments may not be a good indicator of single pressures. They may be related to certain and cumulative pressures and may be indicative and integrative of some form of pressure, but they cannot be related to specific pressures because most anthropogenic activities lead to changes in sediment properties.

Analysis of redox potential provides a rapid means of assessment of the degree of oxygenation within the sediment and can be used as an indicator of organic pollution or enrichment. In intertidal areas, simple field measurements can be carried out to determine the depth of the redox potential discontinuity. Alternatively, more detailed measurements of redox potential can be made using a simple electrode although this can be difficult in unconsolidated sediments. Redox potential is sensitive to temperature change so the integrity of samples analysed in the laboratory must be maintained. As this is difficult, laboratory readings may be subject to a degree of error. In subtidal areas, Sediment Profile Imagery (SPI) (e.g. Rhoads & Germano, 1982; Bonsdorff *et al* 1996; Solan & Kennedy, 2002) is used as a measure of both ecological structure and functioning (bioturbation) and of the chemical state of the sediment in terms of the depth of the redox potential discontinuity. Nilsson & Rosenberg (1997) developed the Benthic Habitat Quality index (BHQ) which integrates sediment surface structures (tubes, mounds etc), structures at depth within the sediment (voids, burrows etc) and the redox potential discontinuity. Parameterization of sediment features in relation to the RPD is useful in the assessment of habitat quality and ecological functioning. This index is used in combination with SPI. Additionally, Rhoads & Germano (1986) proposed the Organism Sediment Index (OSI) to measure benthic community response to organic enrichment and oxygen depletion. The OSI is based on four main metrics: dissolved oxygen conditions; depth of the redox potential discontinuity; infaunal successional stage (measured using SPI); presence or absence of sedimentary methane. OSI values can range from -10 to +11, the lowest values being associated with bottom sediments with no dissolved oxygen. Index values are assigned for different RPD boundaries and for different successional stages (azoic, stages 1-3).

Mazik & Elliott (2000) used low density polyester resins to produce burrow casts to demonstrate the impact of petrochemical pollution on bioturbation. However, whilst this technique was visually effective, it was not quantitative. More recently, the use of Computer Aided Tomography (CT scanning) and microCT is being developed as a 3-dimensional tool for investigating ecosystem functioning (e.g. Perez *et al* 1999; Mermillod-Blondin *et al* 2004; Mazik *et al* 2008).

Other chemical analyses range from simple measurements of metal concentration in the sediment (usually the standard ICRCL suite where the concentration of a standard list of metals results from a single analytical procedure, from a single sample) through organic and persistent organic compounds, organometallic compounds, PCB's, halogenated hydrocarbons and radionuclides. The cost of analysis for each class of chemicals is variable and can increase the cost of a monitoring programme significantly if the relevant indicators or parameters are not carefully chosen in line with the activities known to be taking place and their nature. The value of measuring the different chemicals should also be considered. Whilst sediment quality standards have been derived for some classes of chemicals, they are completely lacking for others, as is robust data linking environmental concentration to biological effect. In some cases (particularly where a chemical may be present but not necessarily form a major component of a known discharge or where detailed analysis of individual compounds requires more than one analytical technique), it might be more cost effective to measure bulk chemical classes rather than individual compounds. For example, total PAH or total PCB. Furthermore, it is most useful to measure these indicators in fine, organic rich sediments where they accumulate rather that in coarse, mobile sediments. Hence monitoring should be targeted. At present, methods for the determination of manufactured nanoparticles are lacking and these substances are not monitored under any statutory programme.

Whilst concentrations of contaminants may be indicative of environmental quality at the time of sampling, erosion and deposition patterns, and subsequent resuspension and redistribution of contaminants means that the level of contamination often cannot reliably be related to the discharge which caused it. Nor can temporal trends reliably be established although this does not mean that chemical measurements are not useful. Chemical speciation and bioavailability are determined by the prevailing environmental conditions within the sediments and in the water column (particle size, clay content, organic content, salinity, temperature, pH, redox potential etc). These therefore determine the potential for uptake by organisms and should ideally be measured in association with any chemical measurements.

Finally, Langenberg & Troost (2008) proposed the use of the volume of accidental oil spills, detected by aerial surveillance, as an indicator of the Good Environmental Status aim to maintain contaminant concentrations at levels which do not cause pollution effects. However, in the context of environmental management, it is important to make the distinction between the general decline in the annual volume of spills and the occasional large scale disasters. In the context of this study, this indicator would represent the pressure of hazardous substances on the marine benthic environment.

Overall, the physical properties of sediment are considered to be useful and essential indicators of sedimentary habitat quality. One or more of these parameters is monitored under the Habitats Directive, OSPAR Convention, Water Framework Directive, the Dangerous substances Directive, London Convention, FEPA, Marine Strategy Framework Directive, the Convention on Radioactive Substances Act and the Euratom Treaty. Most can easily be applied to all types of sedimentary environment and are generally simple and inexpensive. In addition, this information will always be needed to explain the variability in the benthic fauna.

3.1.2 Biological indicators – zoobenthos

Indicators of the biological quality of sedimentary environments can be divided into primary or directly measureable parameters and secondary or derived parameters or indices.

a Primary indicators

Due to their general sessile and sedentary nature and their inability to avoid unfavourable conditions, macrobenthic species are sensitive indicators of environmental change and the mainstay of studies assessing impacts on marine ecosystems. Changes in the number of species, abundance and biomass in response to environmental change (natural and anthropogenically induced) are well documented, particularly in relation to organic enrichment or pollution (e.g. Pearson & Rosenberg, 1978). As such, they have become a component of most (if not all) benthic monitoring programmes.

Additionally, examination of the abundance (A/S) and biomass (B/A) ratios can indicate the distribution of the species and the biomass between the number of organisms present. That is, is the community composed of many organisms representing few species (i.e. an indication of dominance), and is the community composed of a large number of small-bodied organisms? Schwinghammer (1988) used size and biomass spectra as an indication of pollution impacts although such measurements are routinely used to assess the health of shellfish populations (e.g. cockles, mussels and oysters) in terms of recruitment, growth and life span. Calculation of percentage dominance (abundance or biomass) and phylogenetic structure can also be useful indicators of changes in community structure over time.

The use of the above parameters in community characterisation and impact assessment is well established and, hence, there are numerous data sets available for comparison. They are also cheap (relatively) and simple to use. However, they are non-specific in that two communities with entirely different species compositions could appear to have the same structure using these techniques (Warwick & Clarke, 1991). Additionally, changes in these parameters may not necessarily be attributable to human impacts and care should be taken when interpreting data in order to avoid confusion between what is an impact and what is simply natural variability. As such, a number of statistical techniques and biotic indices have been developed (section 5.1.2b).

It has long been recognised that the presence of certain species, in significant abundances, can be indicative of environmental change, be it through structural changes to the community due to pollution (e.g. Pearson & Rosenberg, 1978) or through the spreading of introduced or invasive species. Additionally, the tolerance of a large number of marine benthic species to various human impacts has been documented. For example, the Marine Life Information Network (MarLIN) provides an assessment of the sensitivity of a large number of sediment dwelling species against a set of sensitivity assessment benchmarks. It should be noted that the presence or absence of such species should not immediately be interpreted as being indicative of an impact, largely because the level of confidence associated with the assessment of species sensitivity to various pressures is low. However, the absence of a particularly sensitive species from an environment in which it would ordinarily be found would indicate a requirement for close examination. In that respect, the presence or absence of sensitive / tolerant species may be classed as indirectly indicative of impacts.

In contrast, there are a number of species which have repeatedly been associated with the impacts of organic pollution. Pearson & Rosenberg (1978) list a number of soft sediment species which occur in high abundances in areas affected by organic pollution, the most notable being the polychaete *Capitella capitata*. Given that this response is so well documented, these species can be considered reliable indicators of impact, specifically that associated with organic pollution. It should be acknowledged that where indicator species have been identified, they are usually very impact specific and can therefore not be used in isolation. Munari & Mistri (2008) point out that the sensitivity/tolerance approach is ambiguous in that the classification of taxa is often subjective and may vary between scientists and geographic areas. Furthermore, the links between the effects of human activities and changes in the populations of such species is poorly understood.

b Derived indicators

At a simple level, diversity indices can be used as a descriptor of community health and account for the number of species present, their abundance and their relative abundance (i.e. equitability or evenness). Commonly used indices, used both in routine monitoring and in academic studies, include Shannon-Weiner (H’), Margalef’s index of diversity and Pielou’s index of evenness (the way in which the abundance is divided between the species). However, there are a large number of diversity indices potentially available for use. Whilst diversity indices can be simple to measure, widely applicable and easy to communicate, their use has been widely debated (e.g. Washington, 1984). For example, an "equitable" distribution of numbers would not necessarily mean that the ecosystem is not under pressure of any disturbance. For instance, the reestablishment of tolerant species after an episode of acute pollution would result into an average diversity with a high equitability. The assessment would then be biased. Therefore it is also necessary to consider the taxonomic composition of the community.

Furthermore, high diversity is regarded within the Water Framework Directive as equating to good ecological status and, in some countries' implementation of the WFD, high ecological status is defined by Shannon-Wiener diversity values (H') of >4. However, estuaries or transitional waters are naturally low in diversity due to the generally unfavourable environmental conditions to which estuarine species have adapted. As such, using this classification, the ecological status of estuaries would be considered moderate or low when in fact they were functioning normally. Munari & Mistri (2008) also found this to be the case for Adriatic lagoons. This anomaly has been termed the Estuarine Quality Paradox, whereby the natural features of estuaries appear to reflect the polluted nature of other areas (Elliott & Quintino 2007). Hence the use of indicators showing organic and salinity stress will be unsuitable for estuaries and other transitional waters. This danger of the misclassification of quality status indicates that new indicators will be required for estuaries.

The WFD implementation has led to the development of a number of marine biotic indices although it is of note that these largely apply to soft sediments and tend to be of relevance to organic enrichment or pollution only. Examples include AMBI and M-AMBI (AZTI Marine Biotic Index) (Borja *et al* 2000; Borja *et al* 2004) and its derivatives such as the Infaunal Quality Index (IQI, SNIFFER, 2008a) adopted in the UK, Bentix (Biological Benthic Index, simplified from AMBI, Simboura & Zentos, 2002), the Danish Multimetric Quality Index (DKI, WFD, 2007; Borja *et al* 2009) and the Portuguese Benthic Assessment Tool (P-BAT, Pinto *et al* 2009). The BQI (Benthic Quality Index, Rosenberg *et al* 2004) was developed for use in Sweden and several other indicators have been adopted throughout Europe (Table 1), most of which account for some measure of species richness, abundance, diversity and ecosystem functioning either as a measure of functional or feeding guilds or sensitivity to pollution (particularly organic).

The Infaunal Trophic Index (e.g. Codling & Ashley, 1992) was designed to detect the impact of organic pollution based on the proportional representation of species in different trophic categories although this index has largely been replaced by other indices derived for WFD purposes. A number of other indices have been developed in the past and are outlined in Pinto *et al* (2009). Due to the large number and the fact that many were developed in 1980s/1990s and have now been improved, not all of these indicators have been reviewed here.

The WFD implementation emphasises the use of indicators of community structure, such as diversity, ecological group (based on response to organic enrichment) and abundance which are then combined into multimetric indicators (Quintino *et al* 2006). Munari & Mistri (2008) and Quintino *et al* (2006) reviewed the performance of benthic indicators and found each to give a different indication of ecological status. In only a few cases were the results of different indicators in agreement with each other. Furthermore, Reiss & Kröncke (2005) found that the WFD tools and/or the indices they were based on to be seasonally variable as a result of recruitment and that ecological status could range from good to poor depending

upon the season. In particular, the Shannon-Wiener and Hurlbert diversity indices were susceptible and it was recommended that these indices were not used as metrics within WFD tools. They concluded that if there were different and non-consistent responses of the different indices, it would cast doubt in the minds of managers regarding the value of the indices. Additionally, it may also cause confusion over whether or not remediation measures are required. These problems can be minimised by accounting for seasonality in the sampling design, quality assurance schemes and data treatment protocols (e.g. removing juveniles from the analysis). De Jonge *et al* (2006) considered the measure of ecological functioning used in the WFD indicators to be inadequate and therefore the indicators would not be indicative of health in the widest sense. However, it should be noted that WFD tools have undergone significant development since that time (G. Phillips, Environment Agency, pers. comm.).
Pranovi *et al* (2007) stated that marine biotic indices are largely based on the theory of ecological succession (e.g. AMBI, M-AMBI and their derivatives) along an organic gradient, as proposed by Pearson & Rosenberg (1978). A major problem with such indices is that they consider abundances of stress tolerant species which may also be highly tolerant of natural stressors such as the fluctuating conditions in estuaries. As such, low diversity or high numbers of stress tolerant species are not necessarily indicative of an impact (Dauvin, 2007). The above indices were derived from work specifically in subtidal environments. Suitable indicators for coarse sediments, lagoons, estuaries and intertidal areas are still lacking.

A number of indices have been proposed to address this issue, in particular in relation to lagoon systems and other forms of pollution. The Benthic Opportunistic Polychaetes/Amphipods ratio (BOPA, Gomez Gesteira & Dauvin, 2000; Dauvin & Ruellet, 2007) is based on the high sensitivity of amphipods (excluding *Jassa* spp.) compared to polychaetes and has been shown to be effective in relation to PCBs, pesticides, metals and PAH compounds as well as organic enrichment. Similarly, the Benthic Response Index (BRI, Smith *et al* 2001) has proved a useful indicator of chemical pollution. This index was developed in California but could be adapted and tested in UK waters. The indices FINE (Fuzzy Index of Ecosystem Integrity, Mistri *et al* 2007; 2008) and BITS (Benthic Index based on Taxonomic Sufficiency, Mistri & Munari, 2008) have been proposed for non-tidal lagoons in Italy. In particular, FINE incorporates a measure of primary production and may be worth further investigation.

Table 1. Benthic quality assessment methods proposed and/or approved in Europe

Member State and region	**Assessment method / index**	**AMBI**	**BQI**	**Bentix**	**ISI**	**Sensitive/ opportunist**	**ES100**	**Shannon diversity**	**Abundance distribution**	**Density**	**Richness**	**Margalef index**	**Similarity**	**Simpsons index**	**Biomass**	**Feeding guilds**	**Reference**
Estonia (Baltic)	**ZKI** index of zoobenthos community																WFD, 2007; Borja *et al* 2009
Finland (Baltic)	**BBI** Finnish Brackish Water benthic Index																WFD, 2007; Borja *et al* 2009
Sweden (Baltic, Atlantic)	**BQI** Benthic Quality Index																Rosenberg *et al* 2004
Denmark (Baltic, Atlantic)	**DKI** Danish Multimetric Quality Index																WFD, 2007; Borja *et al* 2009
Germany (Baltic)	**MarBIT** Marine Biotic Index Tool																WFD, 2007; Borja *et al* 2009
Germany, France, Spain (Atlantic), Italy, Slovenia (Mediterranean), Bulgaria, Romania (Black Sea	M-AMBI																Borja *et al* 2004; Muxika *et al* 2007
Norway (Atlantic)	**NQI** Norwegian Quality Index																WFD, 2007; Borja *et al* 2009, Josefson *et al* 2009
Netherlands , Belgium (Atlantic)	**BEQI** Benthic Ecosystem Quality Index																WFD, 2007; Langenberg & Troost, 2008; Borja *et al* 2009
UK, Ireland (Atlantic)	**IQI** Infaunal Quality Index																WFD, 2007; Borja *et al* 2009
Portugal (Atlantic)	**P-BAT** Portuguese Benthic Assessment Tool																Borja *et al* 2009; Pinto *et al* 2009
Greece, Cyprus (Mediterranean)	Bentix																Simboura & Zentos, 2002

A number of numerical and statistical techniques are also available for detecting impacts on the sedimentary environment. Warwick & Clarke (1995) proposed the use of taxonomic distinctness as a means of detecting environmental change. This is a measure of the taxonomic spread of species, rather than the numbers of species. Warwick & Clarke (1995), claim that the technique is independent of sample size and sampling effort, it can be used with simple non-quantitative species lists, and there are possibilities of testing for representativeness using permutation tests. Average taxonomic distinctness (AvTD) is a measure of the average degree to which species in an assemblage are related to each other. Variation in taxonomic distinctness (VarTD) is a measure of the degree to which certain taxa are over- or under-represented in samples. For both indices, a simple permutation test of the hypothesis that the species inventory has a taxonomic structure that is representative of the full biodiversity can be constructed. Major advantages of these techniques include their immunity to sampling effort and sample size and the ability to test departure of the index from expectation. These indices are responsive to changes in environmental quality whilst remaining relatively insensitive to differences in habitat type (Cooper *et al* 2008). Because these indices focus on phylogenetic diversity rather than species richness, they can be considered to be more closely related to functional diversity since a phylogenetically diverse community includes a more diverse range of biological traits (Cooper *et al* 2008).

Bremner *et al* (2003) and Marchini *et al* (2008) used Biological Traits Analysis (BTA) to describe ecological functioning of marine benthic communities. BTA incorporates information on species distributions and their biological characteristics to give a summary of the biological trait composition of a community. This provides an indication of the relationship between organisms and their environment and can be used to determine the impacts of anthropogenic activities on marine benthic systems. This is an indicator of functional diversity which can be reduced in environments impacted by human activities.

Graphical techniques include the use of the log-normal distribution (Gray, 1979; 1981) and the use of k-dominance curves (Lambshead *et al* 1983; Magurran, 1988), where cumulative percent dominance (in terms of abundance or biomass) is plotted against the species rank, on a logarithmic scale (Warwick, 1986). Warwick (1986) proposed a variation on the use of these curves whereby the abundance and biomass curves were overlaid on the same graph. This method is known as the ABC (abundance-biomass comparison) method and is based on the assumption that unstressed or stable environments are characterised by one, or few large species, each represented by few individuals. Whilst these species are rarely the numerical dominants in marine or estuarine communities, they are dominant in terms of the biomass. In contrast, stressed communities are characterised by high numbers of short lived r-strategists with small body size, a high reproductive capacity and a variable population size. Clarke (1990) suggested that the distance between the abundance and biomass curves would be indicative of the degree of stress to which a community was subjected and derived a statistic 'W' to describe this. Like all techniques, this also has its drawbacks, particularly in relation to recruitment events (although consideration of seasonality in the sampling design and data treatment protocols such as the removal of juveniles may minimise these effects). For example, the recruitment of high numbers of polychaetes (a natural occurrence) may lead to high abundance but low community biomass, thus falsely indicating stress (Dauer *et al* 1993). For this reason, the technique is not particularly suitable for use in estuaries.

Raffaelli & Mason (1981) proposed the Nematode:Copepod ratio as an indicator of pollution although its validity was widely debated and the technique was criticised for being over generalised (Coull *et al* 1981). Nematodes and copepods are two of the most dominant meiobenthic groups and it was proposed that a high nematode/copepod ratio would be indicative of environmental stress due to the higher sensitivity of copepods. However, there are difficulties in separating the effects of pollution from those of other environmental variables (e.g. sediment granulometry, seasonality) (Warwick, 1981). This is also an example of another technique developed to detect the effects of organic enrichment which does not perform well in relation to other types of pollution (Lee *et al* 2001).

3.1.3 Biological indicators – marine angiosperms and algae

The Water Framework Directive identifies marine angiosperms (seagrasses and saltmarsh) as one of the biological quality elements to be used for defining the ecological status of subtidal (seagrasses) areas. Monitoring of angiosperms generally involves recording of spatial extent and distribution, either by in-situ mapping techniques, or by remote sensing techniques, the accuracy of which is continually improving. Other, more habitat-specific indicators are discussed below.

Seagrasses are a particularly useful monitoring tool because of their sensitivity to human impacts (Short & Wyllie-Echeverria, 1996; Foden, 2007). Hence, changes in taxonomic composition shoot density, spatial distribution and spatial extent of seagrass beds (including species of *Zostera* and *Ruppia*) can act as indicators of disturbance (Foden & Brazier, 2007). It is of note that the presence of *Zostera* is seasonal and this should be considered in any monitoring plans and in the interpretation of the data. It should be noted that the absence of seagrass does not necessarily indicate poor ecological status and that determination of reference conditions, changes in which may be indicative of impact, is largely based on expert judgment. Historically, there has not been a national monitoring programme for seagrasses and monitoring on a local scale has therefore involved a variety of methods. Therefore, at present, reliable comparison between areas is not possible (Foden & Brazier, 2007).

With respect to intertidal seagrass assessments, members of the North East Atlantic Geographical Intercalibration Group (NEAGIG) Marine Plants Expert group have agreed a common matrix for allocating status. This matrix combines both losses of species and degradation in the % cover (measured as % cover of seagrass within a quadrat, as shoot counting is not practical in intertidal environment). The intercalibration matrix covers both situations where naturally either two or three species of seagrass are found within either a type or where there are differences within types in specified geographic areas (UKTAG, 2007).

Saltmarsh habitats are also sensitive to perturbation with changes in spatial extent, zonation and species diversity being indicative of disturbance (Best *et al* 2007). Pressures such as nutrient enrichment, physico-chemical changes and the high abundances of macro algae, in combination, can lead to habitat fragmentation and ultimate loss of saltmarsh (Best *et al* 2007). As with many classification tools used for the Water Framework Directive, the saltmarsh classification tool requires determination of reference conditions – areas of saltmarsh showing no signs of significant impacts of Human activity.

Unfortunately, due to historical land claim, coastal defence and flood control, determination of reference conditions according to this criterion is not possible. Therefore, reference conditions are based on those areas where the effects of historical land claim are considered to have stabilised. Any decline against this baseline, beyond natural variation in line with cyclic erosion and deposition patterns, would be considered undesirable (Best *et al* 2007). The elements of the saltmarsh classification tool include spatial extent and distribution, physical structure in terms of creeks and salt pans, vegetation structure (zonation and sward structure) and species composition. Whilst further work is needed to improve the understanding of saltmarsh dynamics, their sensitivity to a number of pressures, particularly habitat modification, is well known. This, together with the simplicity of methods which can be used to characterise saltmarsh habitats, makes them an effective indicator of environmental change.

The Water Framework Directive states that macroalgae are a biological quality element to be used in defining the ecological status of a transitional or coastal water body. In the UK, the spatial distribution, extent and frequency of occurrence of opportunistic algal mats (excessive algal growth) respond to changes in nutrient status and problems of eutrophication, toxic substances and most importantly to habitat modification and general stress (Wells *et al* 2007). These parameters are particularly recorded by the conservation agencies responsible for monitoring the condition status of designated SACs but also form part of the WFD macroalgal monitoring tool. Due to the rapid response of algae to high nutrient concentrations, together with the ease of measurement and the large geographic area over which this effect can be identified, this is considered a useful indicator of the health of intertidal sediment habitats. However, opportunistic algal species occur naturally and their presence, or the absence of other macroalgal species, does not necessarily indicate poor ecological status (Scanlan *et al* 2007).

The opportunistic macroalgae tool uses a multi-parameter index for the purposes of assessing the condition, including the total extent of macroalgal bed; the cover of available intertidal habitat; biomass of opportunistic macroalgal mats; biomass over the available intertidal habitat; and proportion of entrained algae (SNIFFER, 2008b). Overall, the assessment of macrophytes and macroalgae is considered a valuable indicator of environmental change or perturbation due to their responsiveness to change. In particular, the excessive growth of opportunistic algae is a direct and specific pressure indicator (nutrient input) which is also indicative of state and impact. Additionally, there are a variety of reliable and, often, inexpensive techniques which can be employed in monitoring. The accuracy of these techniques is improving over time and, as our understanding of the ecological functioning of these systems improves, there is potential for further development of their use as indicators.

Whilst they are expensive and require a combination of spring tides and excellent weather conditions (principally visibility and no/minimum cloud cover), remote sensing techniques provide a rapid and accurate indication of environmental quality over large areas at a spatial scale which it would not be possible to survey by other means. For example, the Environment Agency have used CASI (Compact Airborne Spectrographic Imager) (Environment Agency, 2007) to determine the spatial extent of opportunistic macroalgal mats in several estuaries in relation to eutrophication studies. This technique was used in the Humber estuary where the opportunistic macroalgal coverage was determined as 0.8% for the whole estuary (Mazik *et al* 2008). Ground truth data, collected during the same week as the CASI flights indicated a total opportunistic macroalgal coverage of 2.4% (Mazik *et al* 2008). Considering the difference in the spatial scale of the two surveys, the agreement of these two values can be considered good, highlighting the accuracy of the technique. In particular, this technique has been refined in recent years and errors (e.g. confusion between diatoms and green algae) are no longer a problem.

3.1.4 Species status indicators (e.g. BAP / OSPAR / Red List species)

Species status indicators applicable to marine sediment environments have been derived from International, Europe wide and UK conservation lists. OSPAR adopted Annex V to the Convention and the associated Biodiversity Strategy in 1998. A Biodiversity Committee was subsequently established to deliver the Strategy and has embarked on a series of work streams to help achieve this, including listing species and habitats which are threatened or in decline, and for which action is considered a matter of priority (OSPAR list). For marine sediment environments these species include the Native/Flat oyster (*Ostrea edulis*) and the Icelandic cyprine (*Arctica islandica*) (OSPAR, 2008).

Following recommendation from the Priorities Species and Habitats Review Working Group and the Priorities Review Group, the devolved Governments of the UK published the UK list of priority species (and habitats) in August 2007. This list, a result of the most comprehensive analysis ever undertaken in the UK, contains 1150 species that have been listed as priorities for conservation action under the UK Biodiversity Action Plan (UK BAP) (Maddock, 2008). Of the 87 marine species listed, this review found 15 of relevance to marine sediment habitats which have been discussed further. Species of conservation importance, as detailed by the UK Biodiversity Action Plan, the OSPAR Convention and IUCN Red list of Threatened species associated with intertidal and subtidal sediment habitats are listed in Table 2.

Table 2. UK BAP Priority Species (with OSPAR & Red List designations indicated where applicable)

• *Tenella adspersa* - Lagoon sea slug	• *Styela gelatinosa* - Loch Goil Sea squirt
• *Atrina fragilis* - Fan mussel	• *Edwardsia timida* - Timid burrowing anemone
• *Dermocorynus montagnei* - red alga	• *Edwardsia ivelli* - Ivell's sea anemone
• *Cruoria cruoriaeformis* - red alga	• *Arachnanthus sarsi* - Scarce tube-dwelling anemone
• *Anotrichium barbatum* - red alga	• *Pachycerianthus multiplicatus* - Fireworks anemone
• *Lithothamnion corallioides* - Maerl	• *Nematostella vectensis* (Red List) - Starlet sea anemone
• *Phymatolithon calcareum* - Maerl	• *Ostrea edulis* (OSPAR list) - Native or Flat oyster
• *Funiculina quadrangularis* - Tall seapen	

Traditionally, species indicators focus on determining abundance, biomass and the presence and absence of species to indicate community health and conservation stratus of species. A fuller description of these indicators and parameters used to measure these species are already covered under biological indicators - zoobenthos (please refer to section 5.1.2).

Using species which are long lived and slow-reproducing as an indicator, for example the Sea potato (*Echinocardium cordatum*) and the OSPAR listed Icelandic Cyprine (*Arctica islandica*) as opposed to short lived species (crabs), can show whether human pressure is within limits and seafloor integrity is at a level where the structure and functions of the ecosystems are safeguarded and benthic ecosystems, in particular, are not adversely affected. Specifically, these two species have been used by the Dutch authorities as indicators in the Wadden Sea and the Oosterschelde in relation to physical disturbance caused by fishing (Langenberg & Troost, 2008). The bivalve *Spisula subtruncata* has also been used in The Netherlands as an indicator of staple food for herring and the assessment of Good Environmental Status (Langenberg & Troost, 2008).

Where time series data exist, the Living Planet Index (LPI, Loh *et al* 2005; Collen *et al* 2009) provides a useful tool for showing temporal changes in species abundance. This index is based on General Additive or Chain modelling and is presented as simple time series plot. Advantages include the fact that the calculation allows for missing data and can be used with short time series.

3.1.5 Species status indicators – invasive/introduced species

The Convention on Biological Diversity (CBD) defines an invasive alien species (IAS) as 'an alien species whose introduction and/or spread threatens biological diversity' (decision VI/23). This describes the naturalisation and unintended spread of unwanted organisms in areas where they have not previously occurred naturally (Richardson *et al* 2000; Jay *et al* 2003; Pysek *et al* 2004). IAS are plants, animals or micro-organisms outside of their natural geographic range whose introduction and or spread threatens biodiversity, food security, human health, trade, transport and or economic development. They pose the second biggest threat to biodiversity globally, and in certain ecosystems notably islands, the greatest threat to biodiversity. IAS has reached all corners of the globe and impact biodiversity in many ways.

Non-native species are introduced to the UK either accidentally or deliberately by a number of activities including tourism, aquaculture through the import and release of fish and bivalves for commercial purposes in new locations, and shipping in the transport and discharge of ballast water and to a lesser extent by the fouling organisms on the hulls of ships. Langenberg & Troost (2008) suggested that assessing the percentage of treated ballast water would provide a useful indication of the potential for the transport of non-indigenous species, since ballast water is one of the main factors responsible for this. Once established in a new region, non-native species may invade new areas adjacent to the occupied area by natural dispersal, with these species then displacing native organisms by preying on them or out-competing them for resources such as for food, space or both.

Under the CBD, the United Kingdom has an international obligation to address the impacts of invasive non-native species. In 2009, the UK Government published the Invasive Non-native Species Framework Strategy for Great Britain (Defra, 2009).
Of the 19 most invasive non-native species threatening the marine environment, four can be found in/on sediment habitats or threatening species. These include the Wire weed (*Sargassum muticum*), the Chinese mitten crab (*Eriocheir sinensis*), the Pacific oyster (*Crassostrea gigas*) and the Slipper limpet (*Crepidula fornicata*) (JNCC, 2009a).

At a meeting in 2008, five indicators were agreed by the Global Invasive Species Programme (GISP) expert working group. Of relevance to marine sediment environments is the ecological status indicator which records the number of alien invasive species per country. Other indicators which are more high level management tools include:

- a Red List Index (RLI) for impacts of invasive alien species to show the overall impact of IAS on the extinction risk of species globally. This will be a measure of how fast IAS are driving the world's biodiversity to extinction (and how effectively we are mitigating this).
- Recording the trend in national invasive alien species policy. This would demonstrate the number of national policies relevant to IAS concerns has increased through time as countries acknowledge the IAS problem and commit to responding to this threat.
- Record the trends in international invasive alien species policy. This would also be indicative of the number of international agreements relevant to controlling IAS having increased through time, as have the number of countries party to these agreements.
- Global indicator of biological invasion. This is a composite indicator incorporating the invasion status, national and international policy indicators. It simultaneously provides information on the size of the invasive alien species problem and the policy response to it (GISP, 2008).

The Living Planet Index (Loh *et al* 2005; Collen *et al* 2009) detailed in Section 5.1.4 would also apply here.

3.1.6 Biophysical indicators - habitat extent, structure and condition

A number of sediment habitats have been identified as ecologically important and are protected under the Habitats Directive and/or are listed in the OSPAR List of threatened and/or declining species and habitats. As such, there is a statutory obligation to monitor their condition. Examples of such habitats include intertidal and subtidal seagrass beds, saltmarsh, intertidal mud, sand and gravel/mixed sediment habitats, sand banks, maerl, cobble reef and marine landscape features such as estuaries and lagoons.

Monitoring the spatial extent of these features and habitats is a reporting requirement of the Habitats Directive (Davies *et al* 2001). This enables determination of changes in the spatial distribution and extent of habitats which may reflect physical loss, either to land or change to another habitat type, or physical damage in terms of siltation, abrasion or extraction. Acoustic and remote observation techniques are widely used for characterising sedimentary habitats, spatial patterns in habitat distribution and temporal changes in habitat type. Whilst these techniques are comparatively expensive, involving the use of ships or aircraft, they are well validated and provide a rapid means of assessing habitat extent and physical structure over wide areas. Ground truthing is necessary.

In addition to spatial extent, there is also a requirement to monitor the physical and biological structure of these habitats (e.g. Davies *et al* 2001; Langenberg & Troost, 2008). Many of the indicators applied are listed in sections 5.1.1-5.1.5 and, for example, include number of species, total abundance of organisms and abundance of key species, diversity and/or biotic indices, measures of particle size and sediment quality and macroalgal cover. JNCC (2009b) have recently issued guidelines on the identification and assessment of cobble reef habitats which include measures of the proportion of particles >64 mm and whether the sediment is clast or matrix supported, an assessment of reef extent, height and patchiness and an assessment of dominance by epifaunal species. In addition, topographic or (in subtidal habitats) bathymetric measurements are taken as an indication of the prevailing environmental conditions and the effects of activities such as beach recharge or sand and gravel extraction on seabed morphology. In intertidal environments, measurements of sediment accretion and erosion are often made, particularly in relation to habitat creation or restoration schemes. These measurements are either by direct measurement of erosion and accretion (e.g. Mazik *et al* 2007) or by remote techniques such as LIDAR or dGPS mounted on a hovercraft (e.g. Boyes & Allen, 2007; Stockdon *et al* 2003).

Other indicators indentified by Langenberg & Troost (2008) include measures of seafloor integrity, including measures of physical structure and ecosystem structure and function in relation to physical disturbance caused by fishing. Langenberg & Troost (2008) also identified changes in the dynamic coastline length as a potential indicator for alterations in hydrographical conditions. However, they criticised this suggesting that the relationship between hydrography at sea and coastal defence structures was weak. As such, the indicator was not considered useful in assessing changes to hydrographic conditions and certainly not for assessing marine ecosystem health.

The relationship between biological communities and habitat structure has long been recognised and now forms an important tool for the management and conservation of marine habitats. Connor *et al* (2004) produced a detailed classification of marine habitats, or biotopes and this is now widely used in marine monitoring programmes, particularly in relation to the Habitats Directive. Depending upon the nature of and rationale behind the monitoring programme, the identification and mapping of biotopes can be complementary to and, in some cases, can directly replace (following sufficient ground truthing, particularly in sedimentary habitats), some of the more destructive sampling techniques employed to determine the species present, their abundance and biomass. In particular, this is of benefit to ecologically important species such as *Sabellaria spinulosa* and *S. alveolata*, biogenic reef forming species (addressed in theme 6).

Changes in the biotope composition of an area, biotope richness and the spatial distribution of biotopes are all indicative of environmental change, both natural and anthropogenically induced. Furthermore, based on extensive data analysis, specific biotopes and species assemblages have been identified for particular areas or marine sedimentary habitats. Hence, the unexpected appearance or disappearance of a biotope considered to be typical of an area would be indicative of change and would prompt further investigation.

Techniques available for biotope mapping include the use of photography and remote observation techniques (e.g. ROV and acoustic techniques in subtidal areas), and field mapping techniques involving in-situ species identification and estimation of abundance (Davies *et al* 2001). In the case of sedimentary habitats, accurate confirmation of the biotope may also involve core sampling and characterisation of the sediment properties and biological communities in the laboratory.

In summary, biotope mapping and habitat classification allows a rapid assessment of environmental quality or change, which can often be determined in-situ. By reducing the requirement for taking large numbers of samples, the area which can be surveyed is increased. Whilst the field sampling techniques may require a degree taxonomic skill, the data generated are easily communicable to non-scientists. Consequently, biotope mapping is considered to be an extremely valuable indicator but, as with many of the sedimentary/substratum indicators, it will reflect cumulative and often non-specific pressures.

3.1.7 Biomarkers and bioassays

Biomarkers associated with sediment habitats are less numerous than those associated with species inhabiting rock habitats. Environmental stressors, chemical, physical or biological, have both direct (affecting metabolic pathways and biochemical / physiological processes) and indirect (changing food and habitat availability) effects on biota. Therefore, the assessment of environmental quality can be directly linked with the monitoring of indicators at all levels of biological organization.

A subset of bioindicators, known as biomarkers or biological markers, is generally used to indicate exposure of biota to stressors at lower levels of biological integration (sub-cellular to organism). A number of indicators used as assessment tools for the quality of the marine environment have been proposed as potential indicators for the assessment of the marine biodiversity, by members of the BEEP Project (Biological Effects of Environmental Pollution in marine ecosystems). Although no conclusive links between biomarkers and biodiversity (in relation to most benthic organisms) have yet been established, future evaluation of this category of bioindicators as biodiversity-monitoring tools should be made and their potential inter-calibration with the other biodiversity surrogates should be explored. The use of early warning biomarkers is an example of the precautionary principle and it is assumed that any effects at the cellular or biochemical level in organisms will be translated to higher biological levels (population, community and ecosystem level) if left unchecked. However, the concept of environmental homeostasis implies that systems have an inherent ability to absorb change and that more variable systems such as estuaries have an even greater ability to absorb natural and anthropogenic stressors (Ducrotoy *et al* In press).

At present, the use of biomarkers does not form a major part of marine monitoring in sediment habitats although techniques examining burrowing behaviour of *Corophium volutator* and *Arenicola marina* are in use as a means of Direct Toxicity Assessment (Roddie, 1997). Rank (2009) used intersex in *Littorina littorea* and DNA damage in *Mytilus edulis* as indicators of harbour contamination in Denmark. Other potentially useful indicators include benthic foramniferal assemblages recorded in relation to pollution (Frontalini, 2009) and population dynamics of the ascidians *Ciona intestinalis*, *Ascidiella aspersa* and *Styela plicata* (Monniot, 1986). However the latter indicator was deemed unsuitable due to the error associated with its measurement.

Biomarkers are likely to be good for detecting specific impacts, particularly in relation to chemical pollution, and may provide an early indication of stress before impacts are noted at a population and community level. Certain biomarkers, such as detoxification mechanisms, may be indicative of the effects of precise stressors such as heavy metal or trace organics pollution and thus may relate to specific pressures by activities.

4 Evaluation of the effectiveness of indicators against standard scientific and economic criteria

4.1 Criteria used to evaluate indicators

In order to achieve a consistent critical appraisal of all indicators, the indicators for this ecosystem component have been reviewed and scored against the following set of criteria. These criteria have been built into the online indicators database application and the data has been stored electronically.

A *Scientific criteria*

The criteria to assess the scientific 'effectiveness' of indicators are based on the ICES EcoQO criteria for 'good' indicators. The scoring system is based on that employed within the Netherlands assessment of indicators for GES (2008)[3]. A confidence score of 3 – High, 2 – Medium, 1 – Low is assigned for each question. A comment is given on the reasons for any low confidence ratings in the comment box provided within the database. All efforts have been made to seek the necessary information to answer criteria questions to a confidence level of medium or high.

INDICATOR EVALUATION

i. Sensitivity: Does the indicator allow detection of any type of change against background variation or noise:

Score	3	2	1	Confidence
Options	Usually	Occasionally	Rarely	

ii. Accuracy: Is the indicator measured with a low error rate:

Score	3	2	1	Confidence
Options	Usually	Occasionally	Rarely	

If the indicator scores 1 or 2 for question i. or ii. conclude that it is ineffective and do not continue with the evaluation –the indicator will still be stored within the database as considered but will be flagged as 'insensitive, no further evaluation required'

iii. Specificity: Does the indicator respond primarily to a particular human pressure, with low responsiveness to other causes of change:

Score	3	2	1	Confidence
Options	Usually	Occasionally	Rarely	

[3] Langenberg. V.T. & Troost T.A. (2008). Overview of indicators for Good Environmental Status, National evaluation of the Netherlands.

iv. Performance:

For questions 4a-f, if a score of 1 is given, please consider if the indicator is of real use. Please justify (within the report) continuing if a score of 1 is given.

The following criteria are arranged with descending importance:

a Simplicity: Is the indicator easily measured?

Score	3	2	1	Confidence
Options	Usually	Occasionally	Rarely	

b Responsiveness: Is the indicator able to act as an early warning signal?

Score	3	2	1	Confidence
Options	Usually	Occasionally	Rarely	

c Spatial applicability: Is the indicator measurable over a large proportion of the geographical to which the indicator metric it to apply to e.g. if the indicator is used at a UK level, is it possible to measure the required parameter(s) across this entire range or is it localised to one small scale area?

Score	3	2	1	Confidence
Options	Usually	Occasionally	Rarely	

d Management link: Is the indicator tightly linked to an activity which can be managed to reduce its negative effects on the indicator, i.e. are the quantitative trends in cause and effect of change well known?

Score	3	2	1	Confidence
Options	Usually	Occasionally	Rarely	

e Validity: Is the indicator based on an existing body or time series of data (either continuous or interrupted) to allow a realistic setting of objectives:

Score	3	2	1	Confidence
Options	Usually	Occasionally	Rarely	

f Relatively easy to understand by non-scientists and those who will decide on their use:

Score	3	2	1	Confidence
Options	Usually	Occasionally	Rarely	

Thresholds for scientifically poor, moderate and good indicators:

Combine indicator evaluation scores for:

1. Sensitivity
2. Accuracy
3. Specificity
4. Performance

Evaluation Score	Indicator 'Effectiveness' Category
22-27	Good
16-21	Moderate
9-15 OR not all questions completed due to expert judgement not to continue	Poor

Further economic evaluation required - see section B below

B ***Economic criteria***

Having identified the most scientifically robust indicators using the above stated criteria, a further economic evaluation of those most effective indicators (i.e. those falling in the good or moderate categories) is carried out using the criteria stated below.

1. Platform requirements

Score	4	3	2	1
Options	None e.g. intertidal sampling	Limited e.g. coastal vessel	Moderate e.g. ocean going vessel or light aircraft	Large e.g. satellite or several ocean going vessels

2. Equipment requirements for sample collection

Score	4	3	2	1
Options	Simple equipment requirements e.g. counting number of organisms	Limited equipment requirements e.g. using quadrats on the shoreline	Moderate equipment requirements e.g. measuring physiological parameters	Highly complex method e.g. technical equipment operation

3. Amount of staff time required to plan collection of a single sample

Score	4	3	2	1
Options	Hours	Days	Weeks	Months

4. Amount of staff time required to collect a single sample

Score	4	3	2	1
Options	Hours	Days	Weeks	Months

5. Amount of staff time required to process a single sample

Score	4	3	2	1
Options	Hours	Days	Weeks	Months

6. Amount of staff time required to analyse & interpret a single sample

Score	4	3	2	1
Options	Hours	Days	Weeks	Months

7. Amount of staff time required to QA / QC data from a single sample

Score	4	3	2	1
Options	Hours	Days	Weeks	Months

Thresholds for economically poor, moderate and good indicators:

Evaluation Score	Indicator 'Effectiveness' Category
24-28	Good
19-23	Moderate
7-18	Poor

Those indicators which fall within the **'Good'** or **'Moderate'** economic category will then be tagged within the summary database as 'Recommended' indicators. Indicators can also be 'recommended' via expert judgement even if the evaluation of the indicator does not score well enough to be automatically recommended. This judgement will be justified within the report text.

5 Gap analysis – Review of indicators against relevant pressures and important aspects of ecosystem structure and function

5.1 Review of indicators against pressures and identification of gaps

Please refer to the associated spreadsheet 'FINAL Sediment Pressure_APPENDIX 1.xls' which shows the gap matrix for the soft sediment indicators against ecosystem components and pressures. This gap matrix was produced as a tool to aid authors in identifying significant gaps in current or potential indicators i.e. where important pressures on the ecosystem component have no suitable indicators associated with them. All recommended indicators have been prefixed with [R] and the cells containing them are coloured green.

It should be noted that if a single indicator is associated with more than one pressure within the pressures gap matrix, it may mean that this indicator responds to a range of pressures or the synergistic effects of a combination of pressures. Such an indicator would not necessarily be able to detect change which can be attributed to each individual pressure.

The majority of indicators found for intertidal and subtidal sediments were state and impact indicators (also largely indicative of ecosystem structure and/or function) with fewer indicators of pressure being found. However, at least one (and usually several) indicator was found for each pressure relevant to intertidal and subtidal sediments. Those that were found were generally pressure specific and included litter in intertidal and subtidal areas, with parameters such as number of litter items/unit area and type of litter being recorded. Chemical concentrations and redox conditions in sediment are directly indicative of chemical discharge, waste disposal and organic inputs. Particle size distribution and characterisation of the physical properties of the sediment are also pressure indicators which are primarily associated with physical disturbance (smothering, extraction, abrasion, sealing) of the benthic environment, changes in water flow (e.g. as a result of construction work, man-made structures and climate change). However, the physical properties of sediments are influenced by most anthropogenic activities and the pressures they exert and by natural processes. Hence, whilst particle size distribution is a valuable and essential indicator, it is not tightly linked to a particular activity or pressure. Analytical measurements (sieve analysis and laser diffraction) should be accompanied by a physical description and photographs / video footage. This is particularly important in coarse sediments where collecting a representative sample can be difficult. Finally, indicators relating to the spatial extent, distribution and thickness of opportunistic macroalgal mats are a good direct and specific indicator of both the pressure of and impact associated with nutrient inputs. It should be noted that whilst direct inputs of nutrients can be controlled, inputs from the sediment (e.g. resuspension resulting from bioturbation and sediment resuspension) cannot (Sundbäck *et al* 2003).

The following sections detail the pressures of greatest importance to intertidal and subtidal sediment habitats and the state and impact indicators relating to them.

5.1.1 Intertidal sediment habitats

Most pressures associated with human activities are relevant to intertidal sediment. Of particular importance are those pressures associated with climate change and hydrological change which result in changes in water flow patterns, wave exposure and emergence regime. These pressures result in changes to the sediment properties and, hence, the biological communities inhabiting them and potential for nutrient, carbon and contaminant sequestration and cycling. Additionally, changes in emergence regime can result in coastal squeeze in areas which are constrained by infrastructure or sea defences. Effectively this reduces the extent of the intertidal area which impacts upon the species which depend on it. Changes in salinity will also influence benthic community structure, particularly epibenthic species. However, benthic communities (particularly infaunal) are less susceptible to changes in temperature than rocky shore species and marked changes in species distributions have not been noted for intertidal sediments in the same way as for intertidal rock habitats (e.g. Mieszkovska *et al* 2006).

Physical loss and damage are of great relevance to intertidal areas due to activities such as pipeline and cable installation, coastal defence and beach replenishment, tourism and recreation, fishing and shellfish harvesting. These activities cause direct disturbance, abrasion and sediment removal and also indirectly impact upon sediment structure as a result of any hydrological changes they may cause. A total of 28 indicators are currently used to assess the physical loss or change of intertidal sediment habitats.

Whilst most chemical and waste discharge is subtidal, intertidal discharges do occur making contamination by hazardous substances, organic and nutrient enrichment, de-oxygenation and radionuclide contamination highly relevant to intertidal sediments. Furthermore, contaminants discharged into the subtidal environment are dispersed and may accumulate in intertidal areas. Historic discharges have long lasting implications for the chemical quality of sediments, particularly those which are fine grained and rich in organic matter.

Litter is also a problem in intertidal sediments although infaunal and epibenthic invertebrates are less susceptible to this pressure than, for example, marine mammals and turtles. Finally, removal of target and non-target species occurs as a result of fishing activities and, to a lesser extent, bioprospecting, tourism and recreation.

A whole suite of indicators are available to assess the impacts of a variety of pressures within the marine environment on intertidal sediment habitats although the majority of them are non-specific to particular activities or pressures. For example, change in the taxonomic composition of the benthic community may result from organic enrichment, pollution, physical damage, change in salinity or emergence regime (for example). The effectiveness of monitoring can be increased by interpreting the indicators of pressure and state/impact in line with the activities taking place and at what scale. The most important indicators for intertidal sediment habitats, which apply across all pressures (with the exception of atmospheric climate change, radionuclide contamination and visual disturbance (behavioural impacts)) are the presence / absence of characteristic indicator or important species (1245) and taxonomic composition (species present) (1249). These indicators provide an indication of change in the community structure and hence function. Whilst radionuclides are likely to be present in intertidal sediments, no evidence was found in the literature to suggest any impacts on species composition, abundance or biomass. Biochemical indicators are likely to prove a more reliable indicator of the impacts of radionuclides in sediment.

Other important indicators for intertidal habitats are the spatial extent and distribution of habitats (1266) which provides an indication of change over time and whether it is increasing or decreasing in size in relation to the pressures, and the Living Planet Index (1235) which is a very simple representation of change in biodiversity relying on time series data.

The basic indices of species numbers/richness (132), species abundance (641) and biomass (643) play an important role in determining habitat and species status when assessing the overall pressures of climate change; all hydrological change pressures; all pollution and chemical pressures (except radionuclide contamination); all physical loss and damage pressures; and the biological pressure of removing non-target species. These parameters provide a raw value with no manipulation and therefore there will always be agreement between samples and/or monitoring programmes. They are, however, susceptible to sample size and sample volume (particularly in coarse sediments) and seasonality and these factors should be taken into account when spatial and temporal comparisons are made. Two other indicators which feature across most pressure types are species composition (zoobenthos) (1267) & biotopes present (1236).

Diversity indices are commonly used to assess ecological status although they are highly susceptible to seasonality (see section 5.1.2).

Pressures relating to pollution and chemical change are adequately addressed for intertidal habitats with a combination of Water Framework Directive tools for assessing organic and nutrient enrichment including the WFD opportunistic macroalgal tool (596), WFD fucoid extent tool (597), WFD intertidal seagrass tool (606) and the WFD saltmarsh tool (610) and a suite of parameters within the indicators intertidal sediment contamination – metals and organometallic compounds (618; 1321), intertidal sediment contamination - PAH compounds (621) and intertidal sediment contamination - organochlorines (626).

In conclusion, all the pressures identified in the marine environment summarised in Appendix 1 have one or more indicators to measure the impacts on intertidal sediment habitats and species.

5.1.2 Subtidal sediment habitats

Pressures relating to subtidal sediments are largely the same as those which relate to intertidal areas. However, the pressures associated with substratum removal, abrasion and siltation/ smothering are greater due to activities such as sand and gravel extraction, dredged material disposal, benthic trawling, hydraulic dredging, and offshore construction and exploration work.

A total of 37 indicators were recommended for subtidal sediment environments, most of which (with the exception of chemical and organic inputs and non-indigenous species) were not specific to a particular pressure. As such, many of the indicators are similar to those used in intertidal sediments. Exceptions include those specifically associated with cobble reef habitats (reef height, patchiness, distribution and extent, sediment properties and dominance by epifauna) and the WFD tools for assessing ecological quality. The Infaunal Quality Index (IQI) has been adopted in the UK although like most of the WFD indices, it relates primarily to organic enrichment but is being used to detect changes in ecological status in relation to other pressures. A number of studies have demonstrated that AMBI (a major component of the IQI) responds to other, physical pressures (e.g. Muxika *et al* 2005) although these studies do not test its performance in comparison with other indicators. In contrast, other studies (e.g. Ware *et al* 2009) have demonstrated that AMBI does not respond particularly well to the physical pressures associated with aggregate extraction and dredged material disposal. This was attributed to the fact that it is based on the Pearson-Rosenberg model (1978) and that the same patterns may not apply to physical disturbance but also to the high proportion of species not assigned to ecological groups when AMBI is applied to coarse sediments.

Furthermore, these indices are susceptible to seasonality and are not appropriate for coarse sediments, lagoons or estuaries. However, significant work has been undertaken to establish reference conditions and adapt the index for use in these habitats. The effectiveness of the IQI in these habitats is yet to be fully assessed but is entering a stage of validation after which the tool will either be accepted or rejected (G. Phillips, Environment Agency, pers. comm.). Indicators are under development to address these issues (section 5.1.2) and a number of indicators of ecosystem structure and functioning, generally for soft sediments, are also under development and are described in section 6.2.

5.2 Review of indicators against ecosystem structure and function aspects and identification of gaps

Appendix 2 shows the gap matrix for the soft sediment indicators against ecosystem structure and function. A large number of indicators appear to be available to assess ecosystem structure (e.g. number of species, species composition, abundance, biomass, sediment type, spatial distribution of species, biotopes and habitats). However, indicators for ecosystem functioning are lacking. Whilst many indicators relate to various ecosystem functions, they are not direct measures of function themselves.

In particular, there are no indicators for the purification potential of estuaries, delivery of recruiting organisms, propagule dispersal, export of dissolved detritus, direct measures of nutrient exchange and primary production (although the occurrence of algal mats would act as an indirect indicator of these functions). Gas exchange and water quality movement are not considered to be directly relevant to the benthic environment. Similarly, secondary production is only indirectly addressed through measurements of biomass, abundance, species composition and biotopes present. The parameters relating to saltmarsh extent and distribution are indicative of coastal defence provision whilst topographic and accretion/erosion measurements, together with measurements of particle size distribution, would adequately address sedimentation and tidal flow.

There are a variety of indicators relating to bioturbation in soft sediments. Direct measurements include the use of the Sediment Profile Imaging (SPI) camera which provides an excellent simultaneous representation (both qualitative and quantitative) of a number of physical, chemical and biological parameters. In addition, a number of indices have been derived to assess bioturbation in relation to benthic habitat quality (e.g. BHQ, OSI, see section 5.1.1).

6 Conclusions and recommendations

6.1 Database report tables

The list of indicators recommended following the economic and scientific evaluation is presented in Appendix 3. Indicators recommended by the database are given in column C. Those highlighted in blue are those which were recommended by the database and accepted for consideration for inclusion in monitoring programmes. Indicators highlighted in purple are those which may be worth further consideration and testing in the UK. However, since they are currently under development, they cannot be accepted at this stage. Acceptance or rejection of the recommended indicators is presented in column D with a justification of this decision being given in column E. A brief summary of this output is presented below.

6.2 Identification of an effective indicator set

A total of 128 indicators were evaluated, 115 of these were automatically recommended in the database output. Of these, 100 (highlighted in blue) were considered effective and worthy of inclusion in routine monitoring programmes. It is of note that many of those that were rejected were duplicates (i.e. providing the same information) of other indicators which were already accepted.

6.2.1 Indicators of pressure

Twenty-four indicators of pressure were accepted including the presence of litter and physical parameters relating to sediment quality (e.g. particle size distribution, organic content, redox potential measured as depth of the redox potential discontinuity). A number of chemical parameters were also recommended including concentrations of metals, organometallic compounds, PAH compounds, organochlorines, PCBs and radiocuclides in sediment. It should be noted that contaminants in the marine environment tend to concentrate in organic rich, fine sediments. Monitoring should be targeted in these areas, and also in areas subject to direct (i.e chemical discharge or waste disposal) and indirect inputs (dispersion and concentration of contaminants due to prevailing current patterns). Due to the expense of such monitoring, it is not recommended that these indicators are used for the sake of it. Similarly, the purpose of monitoring should be considered. Many analytical processes generate an output for a whole suite of compounds from one analysis. Where this is not the case, the value of additional analyses should be considered (e.g. are the standard 7 CB congeners sufficient or is there value in more detailed analysis) and the indicators should be chosen on a site by site basis. Other indicators of pressure included simple measures of salinity, temperature, topography and bathymetry and tidal conditions.

Ballast water is considered to be a major source of non-indigenous organisms. Recording the percentage of treated (i.e. to kill such organisms) ballast water has been recommended as a means of assessing this pressure.

6.2.2 Indicators of state and impact

Seventy-six indicators of state and impact were accepted which included primary and derived metrics relating to the zoobenthos, indicators relating to marine angiosperms and algae, indicators relating to species and habitat status and a number of biomarkers.

a Zoobenthos

Simple measures of species richness, taxonomic composition and abundance are recommended as these are primary metrics recorded as part of any routine sampling programme. They are raw values which have not been manipulated and so are comparable on a spatial and temporal scale. They form the basis of most ecological quality and diversity indices, many of which give conflicting results. However, these metrics are affected by sample size, sampling effort and gear type. They are also susceptible to seasonality and so these factors should be considered when any comparisons are made between data sets. Whilst the use of diversity indices has been widely debated, it is recommended that some measure of diversity is included in any assessment of the marine benthic environment. Biomass is a useful measure although care should be taken since the presence of a single large mollusc (for example) can skew a data set. As such, biomass for the major families should be recorded rather than total biomass or biomass of major taxonomic groups (molluscs, annelids etc).

The Infaunal Quality Index (IQI) has been accepted within Europe for WFD monitoring in the UK and incorporates a measure of ecological functioning. For this reason, it has been accepted here. However, it is emphasised that its use should be restricted to the environment for which it was designed – subtidal soft sediments, until the performance of the index has been validated in other sediment types. Many species in coarse sediments have not yet been assigned to ecological groups and the index is therefore not appropriate in such environments. Furthermore, this index, and all other indices which are based on AMBI, is based on the classic response of benthic organisms to organic pollution (Pearson & Rosenberg, 1978). Whilst it has been used in relation to other forms of pollution, it is certainly not an optimum indicator for detecting the impacts of physical disturbance at present. Whilst studies have been undertaken to assess the performance of AMBI (and indices based upon it) in different sediment types and in relation to different pressures, the results have been contrasting (e.g. Muxika *et al* 2005; Ware *et al* 2009). This index is also susceptible to seasonality (although the importance of this in relation to indicator performance is yet to be established) and this should be considered when temporal comparison is made between data sets.

b Marine angiosperms and macroalgae

A large number of parameters relating to the condition of saltmarsh and seagrass beds have been recommended, many of which form part of WFD monitoring tools. They are simple to use and effective and include measures of plant species composition, density, disease, presence of other features which may have negative impacts (e.g. opportunistic macroalgae) and physical structure of the environment (topography, creek morphology). Measurements of the spatial extent, frequency of occurrence, biomass and depth of opportunistic macroalgae are excellent indicators of both the pressure of nutrient input and the impacts.

c Species and habitats

Spatial extent and distribution of characteristic species / species of conservation importance, habitats and biotopes were all recommended as valuable indicators although it was noted that biotope classification can sometimes be subjective and that further development of an objective method of assigning biotopes would be useful. In addition to these parameters, the zoobenthos and macroalage / marine angiosperm indicators are also of value in assessing the status of habitats. The percentage of protected habitats in a given area is considered to be a useful response indicator but does not provide any indication of environmental quality. Nevertheless, the information is easy to obtain and would be valuable as supplementary information.

The Living Planet Index provides a simple useful representation of temporal changes in the status of a species, be it a characterising species, species of conservation importance or a non-indigenous / invasive species. This calculation can also be applied to other metrics o show, for example, change in number of species, abundance, biomass or diversity over time or change in pressure such as chemical contamination. A number of other indicators specific to non-indigenous species were also recommended.

d Biomarkers

Biomarkers do not currently form a major part of environmental monitoring in relation to sediments, although they are considered to be valuable early warning indicators. Four biomarkers were recommended for consideration although, in most cases, further development is required. These include DNA damage in *Mytilus edulis*, imposex in *Littorina littorea*, species composition of benthic foramnifera and a number of reproductive assays.

6.2.3 Indicators of ecosystem functioning

Indicators of ecosystem functioning are currently lacking. The use of the Sediment Profile Imaging (SPI) camera has been recommended as a valuable indicator as it provides information on the physico-chemical conditions within the sediment (principally the redox conditions) together with information on bioturbation, depth of activity and the ecological groups present (in terms of their mode of feeding and bioturbation). This can be used in combination with indices such as BHQ (Benthic Habitat Quality) or OSI (Organism Sediment Index) although it is not considered necessary to measure both. Biological Traits Analysis (BTA) is also considered to be a useful measure of ecological functioning and can be derived from routine sampling data. A measure of change in functional diversity may be more a valuable indicator of change than simple measurements of species richness and abundance.

Measures of spatial extent of macroalage and marine angiosperms are indirect measures of primary production, as is macroalgal biomass. However, no direct measure of primary production is currently in use. Similarly, macrofaunal biomass only provides an indirect measure of secondary production.

6.3 Recommendations for areas of development to address significant gaps

- Quality indices for estuaries, lagoons and coarse sediment habitats are currently lacking. In Italy, several indices specifically designed for lagoons are under development and/or proposed for inclusion in WFD monitoring (e.g.

FINE, Mistri *et al* 2007; 2008). Further investigation and trials in UK lagoon systems are recommended.

- Other indices considered worthy of further investigation include the Benthic Opportunistic Polychaetes/Amphipods ratio (BOPA, Gomez Gesteira & Dauvin, 2000; Dauvin & Ruellet, 2007) and the Benthic Response Index (BRI, Smith *et al* 2001) since these indices were developed and have been applied to forms of pollution other than organic enrichment.

- Measurements of primary and secondary production are currently not included in monitoring. As such, there are no direct measures of these components of ecosystem functioning. For marine angiosperms, macroalgae and diatoms on the sediment surface, simple chlorophyll-a measurements would be indicative of the rate of primary production. Crisp (1984) proposed methods for the determination of secondary production. However, it is anticipated that these measurements, in combination with all the other necessary components of a monitoring programme, could prove impractical and prohibitively expensive.

- Further development of biomarkers to act as early warning systems is recommended. These indicators exist for many other species (particularly fish) and could be of great value to sediment habitats.

- Methods for the determination and an understanding of the behaviour of engineered nanoparticles are currently lacking and monitoring of this class of chemicals consequently does not take place. Further work is needed here since the importance of these particles in an industrial context is increasing.

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