Science Series
Aquatic Environment Monitoring Report **no. 59**

Monitoring of the quality of the marine environment, 2004-2005

Cefas

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Foreword

Aquatic Environment Monitoring Report No. 59 collects together work carried out in 2004-05 by Cefas scientists in support of our monitoring and surveillance duties. (see overleaf). The information presented covers both environmental surveillance at offshore and coastal sites and site-specific work carried out in support of risk assessments and regulatory procedures. Some of the science reported here forms part of wider efforts to integrate data from Departments and Agencies in the UK to provide a comprehensive picture of the quality of the marine environment via the UK Clean Seas Environmental Monitoring Programme (CSEMP). Other components are unique to Cefas due to our requirement to understand ecosystem response resulting from potential pressures from deposit, extraction and discharge activities.

The strategy for the CSEMP is described in publications commissioned by the Marine Environment Monitoring Group (MEMG). The programme manual, known as the Green Book, is available in downloadable format from the Scottish Environmental Protection Agency's website at:

www.sepa.org.uk/marine/

The programme seeks to develop time trend data for a limited number of sites around the UK and this work is augmented by special surveys of compounds likely to pose specific risks, or for which few data exist.

The Defra report Safeguarding our Seas (2002) set out a vision for "clean, healthy, safe, productive and biologically diverse seas". It started a process which has changed the UK's approach to monitoring and assessment of our seas. The next stage in this process was the preparation of the first integrated assessment of our seas, Charting Progress (2005). This provided a baseline for the state of our marine environment at that time, and much was learnt from the process of its preparation. Charting Progress outlined a number of actions, including the development of a UK Marine Monitoring and Assessment Strategy (UKMMAS). Within this strategy, three evidence groups have been established to collate data on the themes of:

- Clean and Safe Seas
- Healthy and Biologically Diverse Seas
- Productive Seas

So as to make explicit the links between the topics covered in this report and the aims of the UKMMAS, the topics have been grouped under these headings. Additionally, major findings and policy implications have been highlighted at the end of each topic.

This report, earlier reports in the AEMR series and other publications are also available in downloadable format from the Cefas website

www.cefas.co.uk

Robin Law
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Background to the work

As an Executive Agency of the Department for Environment, Food and Rural Affairs (Defra), Cefas carries out work in support of Defra's five strategic priorities, all of which underpin the overarching aim of promoting sustainable development:

- Climate change and energy
- Sustainable consumption and production
- Natural resource protection
- Sustainable rural communities
- A sustainable farming and food sector, including animal health and welfare.

Within these priorities, environment work at Cefas is directed at research, monitoring and assessment of the impact of potentially harmful substances or activities on the quality of the marine, coastal and estuarine environments. We are involved directly in advising on UK and international legislation and in developing policy relating to management of the aquatic environment. We provide advice to Governments, enforcement agencies and policymakers throughout the world on the development and implementation of monitoring and assessment programmes and control measures.

An important component of our work is to provide advice to Defra Ministers and other Government Departments on all aspects of non-radioactive contamination of the aquatic environment. Specifically under Part II of the Food and Environment Protection Act (1985) (FEPA) (Great Britain Parliament, 1985a), Defra has the responsibility to licence and control the deposit of material to sea. Following the cessation of the disposal of sewage sludge to sea, licensed materials are predominantly sediments, derived from maintenance and capital dredging activities in coastal waters. Disposal at sea is also regulated internationally by OSPAR, and our work enables the UK to fulfil its obligations as a Contracting Party.

The Cefas Inspectorate evaluates scientific and technical aspects of licence applications and makes regular visits to licence holders to ensure that any stipulated conditions are being met. Conducting monitoring programmes in support of risk assessments enables Defra to ensure the effectiveness of the assessment process and provides a basis for decisions on future policy for the management of marine resources. Cefas scientists monitor the environmental conditions at marine disposal sites and compare the results with those obtained during more general monitoring studies, allowing action to be taken if unexpected impacts should occur. This also provides a feedback loop which ensures that risk assessments undertaken within the licensing process incorporate the most recent research findings.

Under the Water Resources Act (1991) (Great Britain Parliament, 1991), Defra is a statutory consultee for all discharges to controlled (tidal) waters. Cefas scientists assess the fishery implications of applications for consent to discharge permits. Consideration is given to resources in the area, the toxicity of the effluent, local hydrographic conditions and any standards set out in national policy or EU Directives.

We also provide advice to the Department of Trade and Industry (DTI) and the Office of the Deputy Prime Minister (ODPM) concerning the control of pollution in other areas affecting the marine environment including the extraction of offshore oil and gas and marine aggregate. The statutory Offshore Chemical Notification Scheme and the Government View on the winning of aggregates, respectively, control these activities, and the regulatory regime for aggregates is presently also changing to a statutory scheme.

On Defra's behalf, Cefas is responsible for monitoring intermediate and offshore stations within the UK CSEMP, which seeks to integrate national and international monitoring programmes for all UK agencies. Each year, we collect samples of seawater, sediment and biota for chemical analysis and deploy a number of biological effects techniques, including water and sediment bioassays and fish disease surveys. The current phase of the CSEMP is focused on the detection of long-term temporal trends in contaminant concentrations and the development and deployment of a wider range of biological effects techniques studying organism response at a variety of cellular and sub-cellular levels. The CSEMP allows us to ascertain the effectiveness of regulatory measures taken to reduce the inputs of hazardous substances to UK seas. In addition, it contributes to the UK's international monitoring obligations to demonstrate UK compliance with various EU Directives: Dangerous Substances Directive (76/464/EEC); Shellfish Waters Directive (79/923/EEC); Shellfish Hygiene Directive (91/492/EEC); Fishery Products Directive (91/493/EEC); the Commission Decision 93/351/EEC concerning maximum mercury limits in fishery products, and similar requirements under OSPAR. Currently, a group led from within Cefas is working to redesign the CSEMP so as to ensure that it meets current requirements and, as far as possible, to dovetail with proposed monitoring to be undertaken under the EU Water Framework Directive (2000/60/EC) in rivers, estuaries and coastal waters.

In order to ensure that the advice provided to Defra and other regulators is always based on the most up-to-date knowledge and techniques, Cefas carries out a wide range of research and development to provide for the future needs of monitoring and surveillance programmes. For example, we have developed new and more sensitive bioassay techniques, analytical methods and unattended sampling and monitoring devices. Within these programmes we have made a number of significant contributions to environmental protection and as a consequence of our work have established a worldwide reputation in the field of aquatic environmental research. More information on our research programmes is available on the Cefas website: www.cefas.co.uk.

Main findings and their relevance

Clean and safe

1 Radioactivity in UK coastal waters:

- The UK goverment is committed to implementing the OSPAR's 1998 Strategy for Radioactive Substances.
- The survey work carried out here monitors progress towards the 2020 target of reducing concentrations close to zero for artificial radioactive substances in the marine environment.
- Levels of ^{137}Cs are exhibiting a slow decrease with time (eg falling by half about every 6 years in the Irish Sea), whereas tritium concentrations appear broadly constant within the inevitable data scatter.

2 Radionuclide concentrations in dredged sediments:

- Defra undertakes assessments for assurance that there is no significant foodchain or other risk from the disposal of dredge material, known to be contaminated by radionuclides, prior to the issue of licences to operators under the Food and Environment Protection Act, 1985.
- Assessments for dredging at Maryport Harbour and the port of Silloth, contaminated by the legacy of historically high discharges from Sellafield indicated that the impact of the disposal operation was below the '*de minimis*' criteria.

3 The toxicological impacts of oil and chemically dispersed oil: UV mediated phototoxicity and implications for environmental effects, statutory testing and response strategies:

- Defra are currently undergoing a review of the UK oil spill treatment product scheme under which a wide range of testing issues will be investigated. The significance of UV mediated toxicity in the testing process is one of the items under consideration. These preliminary results have been fed into the process.
- Defra have a responsibility for approving the operational use of oil spill dispersants in UK marine waters. Currently the UK National Contingency Plan does not take account of UV levels in guidance about the decision to spray or not. This work may ultimately feed into that process.
- In the future Defra will need to become more involved in the drafting of international policy of the testing and use of oil spill dispersants. The issues of UV mediated toxicity are more acute in areas of higher incident sunlight and clearer waters therefore the findings from this work has a broad international context.

4 Use of passive sampling for environmental monitoring:

- POCIS sampling offers improved assessment of aquatic organisms' exposure to contaminants as a result of integrated sampling.

5 Tri-butyl tin (TBT): a survey of imposex in dogwhelks (*Nucella lapillus*) in the UK and measurement of TBT in water and sediments in areas associated with high shipping activity:

- Wild whelk populations are still adversely affected by TBT from ships' antifouling paints, underlining the importance of the IMO ban and its use on large vessels.

6 A proposed Marine Quality Index for the integration of chemistry, biological effects and biological community data obtained in the UK National Monitoring Programme:

- A Marine Quality Index has been developed which, when applied to UK monitoring data, provides an illustrative method of indicating the relative environmental quality at monitoring sites.

Healthy and biologically diverse

7 Advice on fishery implications of pipeline discharges:

Within general considerations of impact on the marine environment the specific considerations for fisheries are:

- Improvement and protection of shellfish waters and growing areas
- Protection of fishery products edible by man
- Support for sustainable inshore fisheries

8 Contaminants in marine mammals:

- Although the production and new use of PCBs was banned ca. 30 years ago, environmental levels are still high enough to elicit effects and continuing efforts to minimise inputs are warranted.
- Use of the brominated flame retardant HBCD has led to rapidly rising concentrations in the blubber of marine mammals. Continued study is warranted, as these data feed directly into an ongoing EU risk assessment of the continued production and use of this compound.

9 Fish disease investigations 2004 and 2005:

- Multivariate analysis of disease data has allowed sites to be classified according to disease status and provides a tool for integration with biomarkers, environmental and contaminant data in line with ICES/OSPAR initiatives for integrated monitoring.
- Pathology in plaice and dragonet show that they are susceptible to contaminant effects.
- Hyperpigmentation in dab from the North Sea in particular continues to show an upward trend. Investigations into the cause are urgently required.

10 Eastern English Channel broadscale mapping:

The provision of habitat maps in the central Eastern English Channel from this programme will provide several specific purposes, all of which are relevant to Defra policy requirements. These maps will allow us to:

- Place into a wider regional context the aggregate license application areas of the East Channel Region;
- Distinguish areas of potential Annex I reef habitat (bedrock and boulder and cobble fields) under the EU habitats Directive from areas of finer gravel and sand;
- Provide a base map of the physical and biological resources of the area to enable better consideration including monitoring, of potential effects of a range of anthropogenic activities, and to facilitate integrated management of such activities within the area;
- Identify areas of biodiversity interest for consideration as marine protected areas at a national level or under OSPAR Annex V.

Productive

11 Licensing of deposits in the sea:

- Monitoring the quality and quantity of dredged material disposed of at sea and at disposal sites is undertaken to protect the marine environment.

Glossary of terms

AMP	Asset Management Programme
ANOSIM	Analysis of similarities
AQC	Analytical Quality Control
ASG	Ammonium duodeca-molybdophosphate on silica gel
BECPELAG	an ICES/IOC International Sea-going Biological Effects Workshop in 2001
BEI	Biological Effects Index
BFR	Brominated flame retardant
BGS	British Geological Survey
BNFL	British Nuclear Fuels Limited
CB	chlorobiphenyl
CCI	Chemical Contaminant Index
CSO	Combined Sewer Overflows
CEMP	Co-coordinated Environmental Monitoring Programme (of OSPAR)
COPR	Control of Pesticides Regulations
CSEMP	Clean Seas Environment Monitoring Programme
DBT	Dibutyl Tin
DDT	Dichlorodiphenyltrichlorethane
DNA	Deoxyribose nucleic acid
DTI	Department Trade and Industry
EAC	Environment Assessment Criteria
EARP	Enhanced Actinide Removal Plant (at BNFL Sellafield)
ECR	East Channel Region
EDCAT	Endocrine Disruption in Catchments
EEC	Eastern English Channel
EHS	Environment and Heritage Service
EROD	Ethoxyresorufin-O-deethylase
EU	European Union
FEPA	Food and Environment Protection Act 1985
FRS MLA	Fisheries Research Services Marine Laboratory, Aberdeen
GC-FPD	Gas Chromatography with Flame Photometric Detector
GIS	Geographic Information System
HBCD	Hexabromocyclododecane
IAEA	International Atomic Energy Agency
IMO	International Maritime Organisation
JAMP	Joint Assessment Monitoring Programme
LC/MS	Liquid chromatography/mass spectrometry
MALSF	Marine Aggregate Levy Sustainability Fund
MDS	Multi-dimensional scaling
MEMG	Marine Environment Monitoring Group
MEPF	Marine Environment Protection Fund
MFA	Marine Fisheries Agency
MQI	Marine Quality Index
NCP	National Contingency Plan
ODPM	Office of Deputy Prime Minister
NMMP	UK National Marine Monitoring Programme
NOECs	No-effect concentrations
OFWAT	Office of Water
OEB	Oyster Embryo Bioassay
OSPAR	Oslo and Paris Commission
PAH	Polycyclic aromatic hydrocarbon

POCIS	Polar Organic Chemical Integrative Sampling
PRIMER	Plymouth Routines in Multivariate Ecological Research
REA	Regional Environmental Assessment
SERAD	Scottish Executive Environment and Rural Affairs Department
SFPA	Scottish Fisheries Protection Agency
SIMPER	Similarity Percentages Routine
SPE	Solid Phase Extraction
SIXEP	Site Ion Exchange Effluent Plant (at BNFL Sellafield)
SPMD	Semi-permeable membrane devices
STW	Sewage treatment works
TBBP-A	Tetrabromobisphenol-A
TBT	Tri-butyl tin
THORP	Thermal Oxide Reprocessing Plant (at BNFL Sellafield)
UK	United Kingdom
UKMMAS	UK Marine Monitoring and Assessment Strategy
UV	Ultra-violet
UVB	Ultra-violet B type radiation
VDSI	Determination of the Vas Deferens Sequence Index
WAF	Water accommodated fractions
WFD	Water Framework Directive (of the EU)

Clean and safe

1. Radioactivity in UK coastal waters

Author: David McCubbin

1.1 Introduction

The UK government is committed to preventing pollution of the marine environment from hazardous substances, which includes ionising radiation. The ultimate aim is to reduce concentrations in the environment to close to background values for naturally occurring radioactive substances, and close to zero for artificial radioactive substances (Defra, 2002). A long-term programme of surveillance into the distribution of key radionuclides is maintained using research vessels, and other means of sampling. The results obtained from the seawater surveys reported here provide evidence of progress towards achievement of the Government's vision. Summary data is also set out in a recent report (Marine Environment Monitoring Group, 2005). In addition, these surveys support international studies concerned with the quality status of coastal seas (eg OSPAR, 2000) and provide information that can be used to distinguish different sources of man-made radioactivity (eg Kershaw and Baxter, 1995).

Detailed historical data for ^{134}Cs and ^{137}Cs in seawater have been published in a series of reports so as to aid model development (Camplin and Steele, 1991; Baxter *et al*., 1992; Baxter and Camplin, 1993a-c) and have been used to derive dispersion factors for nuclear sites (Baxter and Camplin, 1994). The data have also been used to examine the long distance transport of activity to the Arctic (Kershaw et al., 1999) and long-term trends in Northern European seas (Povinec *et al*., 2003).

Discharges from Sellafield peaked in the mid 1970s. A number of counter-measures were introduced, including the Site Ion Exchange Effluent Plant (SIXEP, in 1986), which controlled Cs discharges, and the Enhanced Actinide Removal Plant (EARP, in 1994). EARP allowed the treatment of medium-active, stored liquors, which also contained ^{99}Tc - not treated by EARP- and consequently these discharges (which are of limited radiological significance) rose in 1994. However, following a successful trial of new abatement technology, discharges once again decreased in 2003 (Mayall, 2005). Discharges of ^{129}I, ^{90}Sr, ^{14}C, ^{60}Co, and ^{3}H also increased in the mid 1990s, as a result of operational changes at the site, including the starting up of the Thermal Oxide Reprocessing Plant (THORP) in 1995.

Studies of the migration behaviour of ^{99}Tc have afforded opportunities to substantiate and extend the information obtained from earlier similar studies of ^{137}Cs. The distribution of ^{99}Tc in waters around the British Isles prior to, and immediately after, the increased ^{99}Tc discharges (in 1994) indicated a rapid advection of ^{99}Tc within and from the Irish Sea to the north of Scotland as compared to previous estimates (Leonard *et al*., 1997a,b; McCubbin *et al*., 2002). The subsequent transport rate out of the North Sea and northwards with the Norwegian Coastal Current and West Spitsbergen Current slowed markedly, in apparent correspondence with variations in the North Atlantic Oscillation (NAO) winter index (Kershaw *et al*., 2004).

1.2 Methods

1.2.1 Sampling

The research vessel programme on radionuclide distribution currently comprises an annual survey of the Bristol Channel together with biennial surveys of the Irish Sea and the North Sea. In 2005, coverage was extended to northern Scottish coastal waters with the assistance of the FRS Marine Laboratory (Aberdeen). Large volume surface seawater samples (50 litres) are collected, using the ships pumped supply, during cruises of the research vessels, *Cefas ENDEAVOUR* and *CORYSTES*. Surveys of the Bristol Channel, Irish Sea and the western English Channel were carried out by Cefas in September/October 2005, June 2005 and October-November 2005, respectively. Samples from waters to the west and north of Scotland were collected by staff from FRS Aberdeen between April-August 2005.

1.2.2 Sample analysis

Samples were filtered (0.45 µm) to separate dissolved and particulate phases. Analyses of dissolved ^{137}Cs involved pumping filtered seawater, acidified with nitric acid, through cartridges filled with ASG resin (ammonium duodeca-molybdophosphate on silica gel) to extract caesium. Analyses of ^{3}H involved double distillation of water samples under alkaline conditions and in the presence of holdback carriers to ensure chemical separation from all gravimetric and radiometric interference. Subsamples of distillate were assayed for ^{3}H using a Packard Tri-Carb 2550 TR/LL liquid scintillation counter.

1.3 Results and discussion

The results of the seawater surveys are given in Figures 1.1(a)–1.1(g).

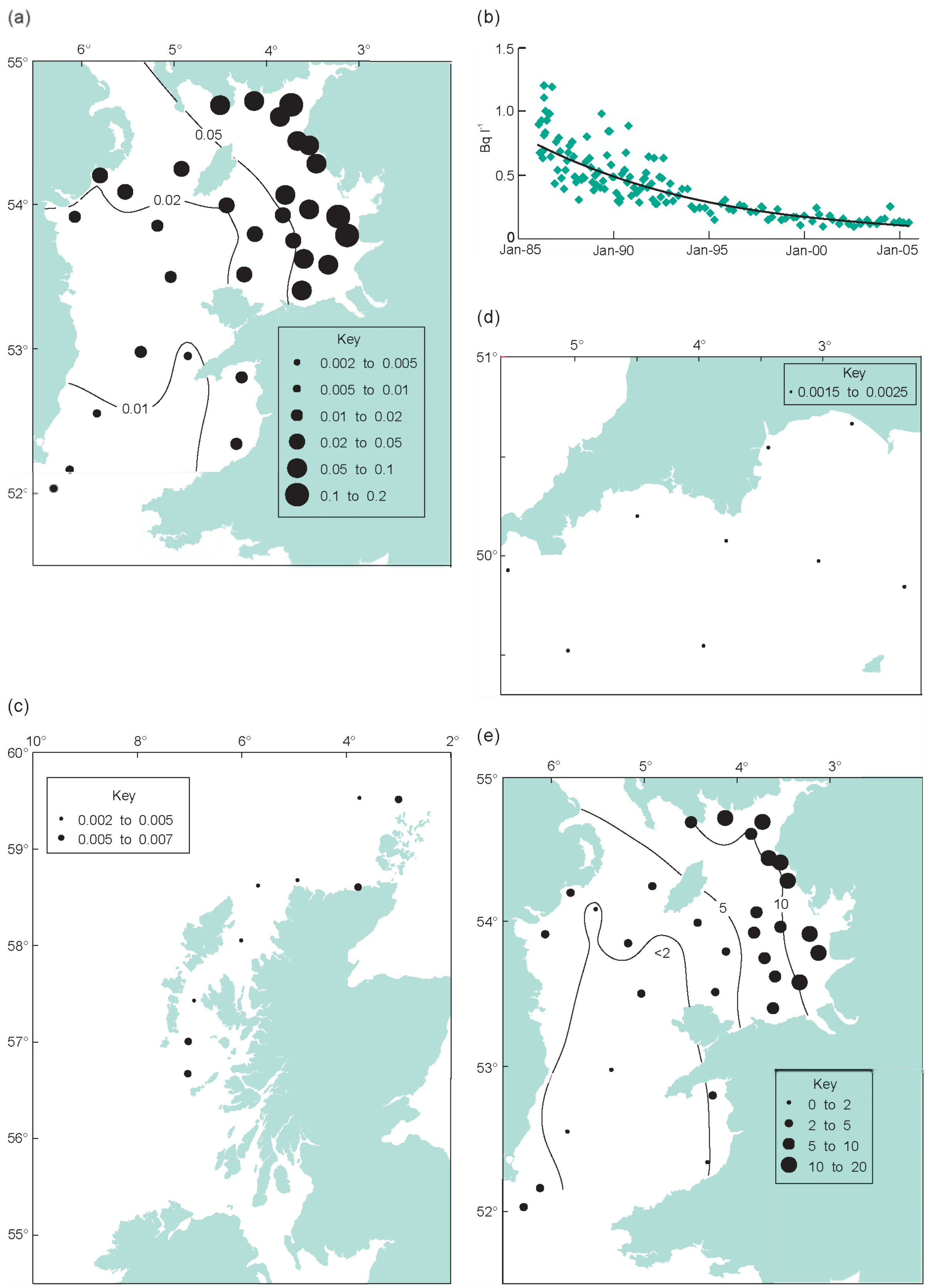


Figure 1.1. Activities (Bq kg^{-1}) of ^{137}Cs and ^{3}H in surface seawater from the UK continental shelf: (a) dissolved ^{137}Cs in the Irish Sea (June 2005); (b) temporal variation of dissolved ^{137}Cs in shoreline seawater close to Sellafield (at St Bees), 1986- 2005; (c) dissolved ^{137}Cs in Scottish waters (April-August 2005); (d) dissolved ^{137}Cs in the western English Channel (October - November 2005); (e) ^{3}H in the Irish Sea (June 2005).

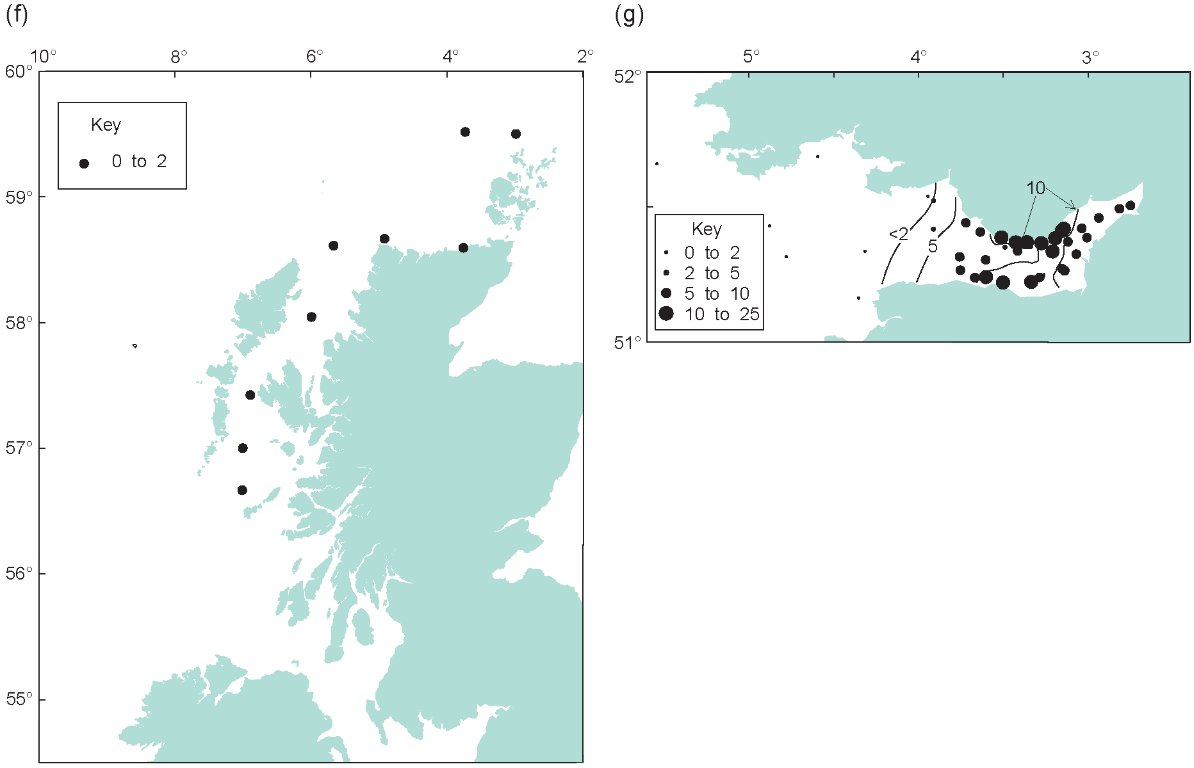


Figure 1.1. continued: Activities (Bq kg^{-1}) of ^{137}Cs and ^{3}H in surface seawater from the UK continental shelf: (f) ^{3}H in Scottish waters (April-August 2005); (g) ^{3}H in the Bristol Channel (September - October 2005).

1.3.1 ^{137}Cs distribution

The Irish Sea ^{137}Cs data (Figure 1.1(a)) indicate that the concentrations observed along a large section of the British coastline, extending from Liverpool Bay in the south to the Mull of Galloway in the north (typically 0.05–0.1 Bq kg^{-1}), were significantly greater than those observed along the Irish coastline (typically 0.005–0.02 Bq kg^{-1}). The ^{137}Cs contours extend parallel to the Cumbrian coastline. The overall distribution of ^{137}Cs is in line with that expected from our knowledge of mean surface water circulation in the Irish Sea (Dickson, 1987). The predominant flow of water is northward via input of Atlantic water from St. George's Channel, passing to the west of the Isle of Man. A minor component of the flow enters the eastern Irish Sea to the north of Anglesey and moves anti-clockwise round the Isle of Man before rejoining the main flow to exit through the North Channel. The ^{137}Cs activities observed here are only a small percentage of those prevailing in the late 1970s. Levels as high as 30 Bq kg^{-1} have been observed in the vicinity of the Sellafield outfall (Baxter *et al.*, 1992) during the period when discharges from Sellafield were substantially greater. Indeed, differences between the ^{137}Cs/^{99}Tc ratio in Sellafield discharges and seawater indicate that ^{137}Cs remobilisation, from sediments contaminated by large discharges in the 1970s, is presently the predominant (~90%) source term to the water column (McCubbin *et al.*, 2002). Consequently, levels in seawater have shown a near exponential decrease with time (ie falling by half about every 6 years) since the commissioning of the SIXEP waste treatment process in the mid 1980s, as illustrated by the data provided in Figure 1.1(b) for shoreline seawater at St Bees (~ 10 km to the north of Sellafield).

The ^{137}Cs data for Scottish waters (Figure 1.1(c)) show low concentrations (less than 0.01 Bq l^{-1}) throughout the survey area. Nevertheless, the concentration in all the samples analysed here remained slightly elevated above the global fallout level now found in North Atlantic surface waters (approximately 0.0012 Bq l^{-1} in 2002 (Bailly du Bois pers. comm.)).

Concentrations in the western English Channel (average activity 0.002 Bq kg^{-1}) were only slightly, if at all, enhanced compared with the background level resulting from global fallout (Figure 1.1(d)).

1.3.2 ^{3}H distributions

Levels of ^{3}H in the Irish Sea (Figure 1.1(e)) were below the limit of detection (~2 Bq kg^{-1}) over a large proportion of the survey area. However, the impact of discharges from Sellafield and the Heysham nuclear power plant was apparent along the Cumbrian and southern Scottish coastline, extending from Morecambe Bay in the south to Luce Bay in the north. Along this section, ^{3}H activities were in the range 10-17 Bq kg^{-1}.

Tritium concentrations in Scottish waters (Figure 1.1(f)) were below detection in all samples.

In the Bristol Channel (Figure 1.1(g)), the greatest ^{3}H concentrations in 2005 (>10 Bq kg^{-1}) were observed in the Severn estuary close to the Welsh and English coastlines. These data indicate measurable elevation in levels close to the Hinkley nuclear power plant and the Amersham radiopharmaceutical plant at Cardiff. ^{3}H concentrations decreased rapidly with distance downstream of the points of discharge (ie, in a westerly direction). Concentrations at the mouth of the Bristol Channel were below the limit of detection (2 Bq kg^{-1}). The spatial distribution is consistent with conservative dispersion behaviour in the macrotidal Severn estuary. Tidal current speeds generally exceed 1.5 m s^{-1} at springs and 0.75 m s^{-1} at neaps, meaning water parcels can move up to 25 km during a flood or ebb tide (Uncles, 1984).

1.3.3 Other radionuclides

Concentrations of ^{99}Tc in seawater are now decreasing, following the installation of new effluent treatment procedures at Sellafield. The results of research cruises involving studies of this radionuclide have been published by Leonard *et al*. (1997a and b, 2001, 2004) and McCubbin *et al*. (2002). Trends in plutonium and americium concentrations in the seawater of the Irish Sea have been considered by Leonard *et al*. (1999). A full review of the quality status of the north Atlantic has been published by OSPAR (2000).

2. Radionuclide concentrations in dredged sediments

Author: David McCubbin

2.1 Introduction

In England and Wales, Defra issues licences to operators for the disposal of dredged material under the Food and Environment Protection Act, 1985 (Great Britain Parliament, 1985a). The protection of the marine environment is considered before a licence is issued. Since dredge material may contain radioactivity, assessments are undertaken where appropriate for assurance that there is no significant foodchain or other risk from the disposal. In 2005, specific assessments of the disposal of dredge material from Maryport Harbour and the port of Silloth were carried out. At both locations, the sediments contain artificial radionuclides due to discharges from BNFL Sellafield and from other widespread sources such as weapon test fallout.

2.2 Materials and methods

Samples of surface sediments were collected from two or three locations to ensure the data provided representative information. Radionuclide assay was achieved using gamma-ray spectrometry by which it is possible to simultaneously measure a wide range of radionuclides commonly found in radioactive wastes.

2.3 Results and discussion

Results from the sediment analyses are provided in Tables 2.1 and 2.2.

The assessments showed that the impact of the radioactivity associated with the disposal operation was very low, below '*de minimis*' levels of exposure. '*De minimis*' relates to doses of the order of 0.010 mSv or less. Guidance on exemption criteria for radioactivity in relation to sea disposal is available from the International Atomic Energy Agency (IAEA, 1999 and IAEA, 2003) and this has been adapted to reflect operational practices in England and Wales (McCubbin and Vivian, 2006).

Table 2.1. Concentrations of radionuclides in sediment from Maryport Harbour, Cumbria, 2005.

Area	Radioactivity concentration (dry), Bq kg^{-1}					
	^{60}Co	^{137}Cs	^{226}Ra (via ^{214}Pb)[1]	^{232}Th (via ^{228}Ac)[1]	^{238}U (via ^{234}Th)[1]	^{241}Am
Senhouse Dock	28	561	27	35	58	930
Elizabeth Dock Approach	18	385	21	25	48	572
Elizabeth Dock	37	930	27	33	47	882

[1] Parent nuclides not directly detected by the method used. Instead, concentrations were estimated from levels of their daughter products.

Table 2.2. Concentrations of radionuclides in sediment from Silloth, Cumbria, 2005.

Area	Radioactivity concentration (dry), Bq kg^{-1}					
	^{60}Co	^{137}Cs	^{226}Ra (via ^{214}Pb)[1]	^{232}Th (via ^{228}Ac)[1]	^{238}U (via ^{234}Th)[1]	^{241}Am
A	9.4	661	23	33	49	581
B	9.9	531	23	24	49	508

[1] Parent nuclides not directly detected by the method used. Instead, concentrations were estimated from levels of their daughter products.

3. The toxicological impacts of oil and chemically dispersed oil: UV mediated phototoxicity and implications for environmental effects, statutory testing and response strategies

Authors: Mark Kirby, Brett Lyons, Jon Barry and Robin Law

3.1 Introduction

The threat of oil-based hydrocarbon contamination via spills and other sources in the aquatic environment and their subsequent treatment with chemical dispersants remains a very real and highly emotive issue. It is well documented that ultra-violet (UV) light can enhance or induce the toxic effects of certain environmental contaminants (termed phototoxicity). Among those contaminants of primary concern are the polycyclic aromatic hydrocarbons (PAHs). Numerous laboratory studies have now demonstrated that the toxicity of PAHs increases 2 to >1000 times in the presence of UV (Arfsten *et al*., 1996; Wernersson, 2003). Pelletier *et al*., (1997) reported that specific components of oil (fluoranthene, pyrene and anthracene) and four oil products were 12 to > 50,000 times more toxic (LC_{50} tests) to marine invertebrate juveniles (*Mysidopsis bahia*) and bivalve larvae and embryos (*Mulinia lateralis*) when compared with parallel tests omitting UV light. Similar studies were conducted using the freshwater crustacean, *Daphnia magna*, to screen for the phototoxic potential of 22 water accomodated fractions (WAF) derived from petroleum based products (Wernersson *et al*., 2003). In this study 16 of the 22 WAFs demonstrated significantly increased toxicity in the presence of UV-light. Lyons *et al*., (2002) highlighted the ability of certain PAHs to exhibit photo-induced toxicity at levels of UV light typically found to penetrate the upper 5 metres of European surface waters. Additional studies, carried out as part of the 2001 BECPELAG ICES/IOC sea-going workshop, demonstrated that sea surface microlayer samples collected from around the Statfjord oil field displayed enhanced phototoxicity to embryos of the Pacific oyster (*Crassostrea gigas*) when exposed to environmentally relevant levels of UV-light (Lyons *et al*., in press). While it is noted that phototoxicity is not restricted to PAHs (the same phenomenon has been demonstrated for other pollutants including pesticides (Ankley *et al*., 1998), munition waste products and metals (Arfsten *et al*., 1994)), it is clear from the literature that this group of ubiquitous contaminants offer the greatest cause for concern for environmental damage.

Even allowing for a growing body of information on UV-mediated toxicity in the literature it remains difficult to identify specific scenarios where information on phototoxicity may be used to modify human activity in order to mitigate potential effects. Phototoxicity has not generally been recognized by scientists and risk managers in the oil and gas industry as of great significance. Traditional toxicological risk assessment studies used to define the hazards of petroleum based discharges to aquatic organisms have generally been conducted inside, under fluorescent lighting which contains minimal UV radiation. In particular, oil spills, and their subsequent treatment with dispersants, offer the potential for significantly increased impacts under conditions of raised UV exposure. Statutory toxicity tests, used as part of the oil spill treatment product approval scheme in the United Kingdom (Kirby *et al*., 1996), are procedures that are carried out in the absence of UV light representative of 'at sea' conditions. It was hypothesised that chemically dispersed oil used in the test procedures may exhibit photoinduced toxicity and a preliminary experiment was conducted in order to investigate this.

Water accommodated fractions (WAFs) of Kuwait crude oil were generated using the standard UK Sea Test equipment (Kirby *et al*., 1996). Briefly, 18 ml of oil was added to 18 litres of seawater (1000 ppm) and then mixed into the water column (mixing propeller set at 800 rpm) for 100 minutes. After a short settling period WAFs were drawn off from a tap at the bottom of the cylindrical tank. WAFs were generated for Kuwait oil only and also with the same amount of oil treated with a representative UK approved dispersant at a ratio to the oil of 1:10.

A range of dilutions (from 1% to 100%) of the 'oil only' and 'dispersed oil' WAFs were tested for toxicity with the oyster embryo bioassay (OEB) under conditions of both standard laboratory fluorescent lighting (lacking UV) and environmentally realistic UV levels using a 12-hour light and 12-hour dark photoperiod. Ultraviolet light was provided by 2 Cleo (Phillips) 20W lamps (3.8-6.3 μW/cm^2 and 280-456 μW/cm^2 UVB and UVA respectively) and light intensities quantified using a spectroradiometer (Glen

Spectra Ltd). Levels of UV light used in this experiment were based on previous field light conditions measured 1 m below the water surface (McCloskey and Oris, 1993). The OEB test is a standard OSPAR bioassay assessing toxicity to the embryo-larval stages of the Pacific oyster, *Crassostrea gigas*, and a recommended method under the Joint Assessment Monitoring Programme (JAMP) for the assessment of water quality in European waters (Stagg, 1998).

The embryo development data was modelled with a model that assumed that the percentage relative normal development reduced exponentially with increasing dose. We can write this model formally as:

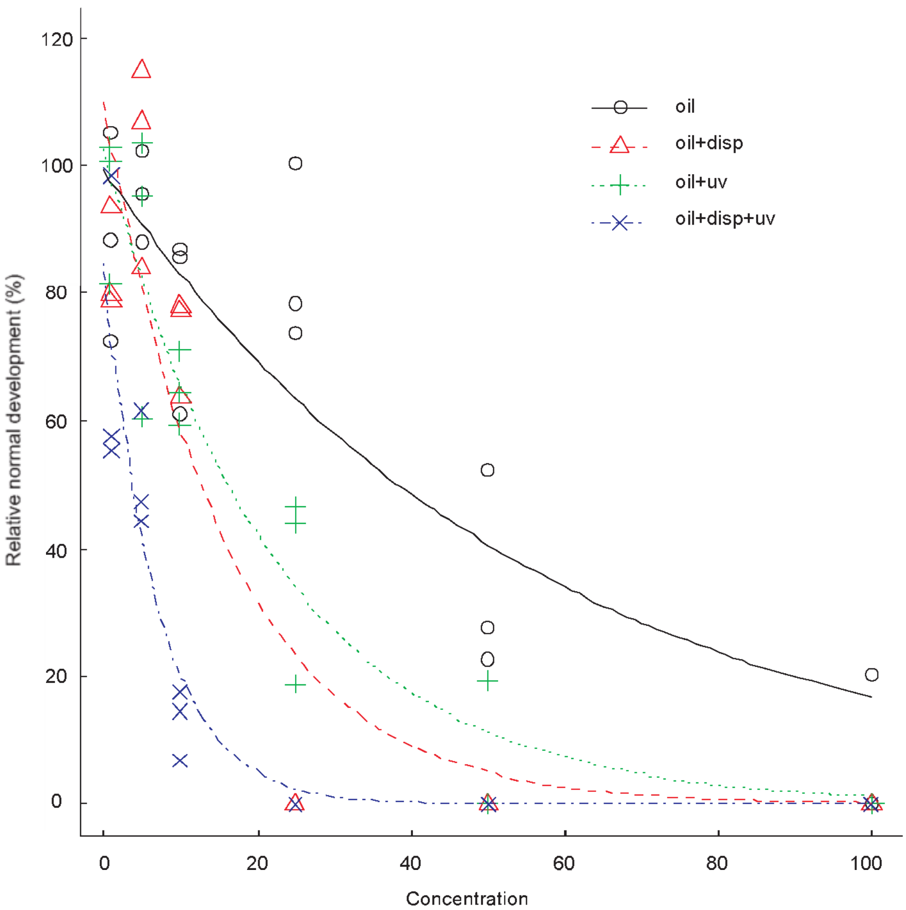
$$E = a \exp^{-b(conc)} + \text{error} \qquad (1)$$

where conc is concentration, E = the percentage relative normal development and a and b are parameters.

The parameters were estimated by minimising the residual sum of squares between the data and the model. We have defined a form of lethal concentration LC_{50} that is the concentration at which the percentage relative normal development is 50%. From the above model, we can estimate this by: $LC_{50} = (\ln(a) - \ln(50))/b$.

The exponential decay model in (1) was fitted for the four treatments and a LC_{50} estimated for each. Figure 3.1 shows the data and the fitted models. We can see that the oil only treatment has the least affect on embryo survival (LC_{50} = 38.4% of the WAF) and that survival is reduced in the presence of UV light (LC_{50} = 16.3). For the dispersed oil, embryo survival falls very rapidly both with (LC_{50} = 3.7) and without UV light (LC_{50} = 12.7). We note that the model fits reasonably well to all of the treatments except, perhaps, for the treatment with oil+dispersant – however, a better fitting model would only marginally affect the estimate of the LC_{50} for this treatment.

Figure 3.1. Oyster embryo development after exposure to water accommodated fractions (WAFs) of mechanically and chemically dispersed Kuwait crude oil under conditions of normal laboratory (no-UV) and UV light illumination conditions. Three replicates for each treatment are plotted and results are expressed as relative normal development compared to the control (no oil dispersant or UV light). Lines were fitted using the exponential decay model[1].



In order to confirm statistically that there was a difference between the four treatments, a randomization in the spirit of Manly (1998) was performed. First the observed residual sum of squares from the total of the residual sums of squares from the four models shown in Figure 3.1. Then 12 observations at each concentration level were randomly assigned to a treatment. The model fitting was repeated and the total residual sum of squares calculated. This randomization was repeated 1000 times and the observed residual sum of squares compared against the randomization distribution, which assumed no differences between the treatments. The observed residual sum of squares of 14,033 was less than any of the randomized values so we can assume that the treatments are different with $p<0.001$.

These preliminary results clearly show that Kuwait crude oil, both mechanically and chemically dispersed, demonstrates significant levels of photo-enhanced toxicity. The mechanically dispersed oil WAF demonstrated toxic effects at 50% dilution under normal laboratory conditions but effects are evident at concentrations as low as 10% under UV conditions. When dispersed oil was tested effects were apparent at 25% and 5% dilutions under the room and UV conditions respectively. Comparisons of the no observed effect concentrations (NOECs) suggest that UV illumination (of an intensity observed in the upper layers of the sea) lowers the concentration of the onset of WAF toxicity of Kuwait crude by up to 5 times and that with dispersed oil the UV mediated effects are at a point approximately 10 times lower. The impact of UV light on WAF toxicity is also borne out by the calculated LC_{50}s with the results showing a 2 and 4 fold increase in toxicity with mechanically and chemically dispersed oil respectively.

These preliminary results have demonstrated the ability of crude oil derived contaminants to exhibit substantial increases in biological effects as a result of UV-mediated phototoxicity. Furthermore, it is clear that the use of chemical dispersants on oil not only increase the toxicity of the WAF but can also augment the magnitude of the UV mediated toxicity. In this study the toxicity of a dispersed oil WAF to molluscan embryos has been shown to increase approximately 10-fold under UV illumination, however, other studies in the literature (Barron *et al.*, 2003) suggest that photo-enhanced toxicity may be much greater for other oil and dispersant combinations and have demonstrated effects on a range of important taxa including crustacea and fish larvae. There is no doubt that there is great potential for UV radiation to increase the impacts of oil derived contaminants (eg PAH) in the upper layers of the marine environment and particularly in shallow water and coastal areas (Peachey, 2004). These sensitive ecosystems are often important environments for the developmental stages of many commercial marine species and studies have previously linked surface water contamination with adverse impacts on fish eggs and larvae in both the North Sea (Westernhagen *et al.*, 1987; Cameron and Westernhagen, 1997) and coastal waters around North America (Hardy *et al.*, 1987). It follows that when these same ecosystems are exposed to an oil spill or to dispersed oil drift an additional risk factor will be the increased threat posed by UV radiation.

There are at least two future issues to consider if one accepts that UV-mediated effects may be significant in the impacts and assessment of oil spills and treatment options. Firstly, is there a need to take account of this issue as part of the statutory toxicity testing protocols for oil spill treatment products? Secondly, does this information have operational implications with respect to dispersant application scenarios and should it be a significant consideration in any net environmental benefit analysis?

The current UK approval scheme for the approval of oil treatment products is based on toxicity assessments conducted wholly in laboratory type conditions, ie, with virtually no UV present. Approvals are based on a comparison of how a product changes the toxicity of oil when applied (Kirby *et al.*, 1996). The premise is that the addition of a dispersant (or other treatment option) should not significantly (allowing for some scientific judgement) increase the toxicity of the oil alone. It could be hypothesised that under 'real' conditions, with UV light present, the dispersed oil may elicit a higher toxicity than are assessed under the current procedures thus leading to an underestimation of the environmental impacts. Although this theory has not been tested it seems unlikely that UV-radiation would significantly impact the results gained from the statutory Sea test for two reasons; i) UV has to be able to penetrate the water sufficiently - during the initial dispersed oil exposure phase the water can be quite opaque and the more likely impact would be during the subsequent 24 hour recovery period in clean flowing water where any accumulated PAHs could be photoactivated; and ii) UV has to penetrate the organism. The brown shrimp, *Crangon crangon*, used in the standard Sea Test can vary in colouring from dark brown to relatively translucent so impacts will be dependent on the nature of the test stock. The UK oil treatment product approval scheme is, however, undergoing a review and any changes to testing regimes to include product only testing, WAF testing or the use of more 'transparent' organisms may need to take account of UV-mediated phototoxic potential.

Phototoxicity may be more appropriate to consider when assessing courses of action during a response scenario. Currently the UK oil spill response National Contingency Plan (NCP) does not take account of any weather conditions other than wind strength/direction and sea state in advising when to potentially spray dispersants. With respect to minimising the impact of an oil spill or dispersed oil plume due to photo-induced toxicity decision makers may also need to take account of; season/time of day/cloud cover (to assess the level of UV light reaching the water surface), water turbidity (to assess UV penetration into the water column) and the nature of organisms in the vicinity and water depth (to assess what ecosystems/ species may be specifically at risk). For example, the depth of water required to remove 90% of UVB from light penetrating surface waters ranges from a few centimetres in eutrophic, turbid waters to 20 m in clear offshore ocean waters (Kirk, 1994; Morris *et al.*, 1995; Williamson *et al.*, 1996; Boelen *et al.*, 1999). Without further research the potential impacts of oil spill treatment from phototoxic activity will remain a matter of speculation but it is certainly worth considering the potential impact difference in a given area between a spill on a cloudless summer day when the water is very clear and that on a cloudy winters day when the water is turbid. Currently the two scenarios would probably be treated the same when deciding what course of spill remediation to undertake.

Finally, the phototoxicity of oil and dispersed oil has global implications. Much of what has been considered in this paper has been in the context of response in northern European, temperate regions. However, many of the busiest tanker shipping lanes and the some of the World's largest marine oil fields (not to mention those yet fully explored or exploited) are in sunnier regions with a much higher average incidence of surface UV-irradiation, clearer waters and more diverse ecosystems. It could be said that it is these areas that are under the most threat from increased impacts. Not to mention the potential future implications of ozone layer depletion and global warming!

4. The use of passive sampling for environmental monitoring

Authors: Jan Balaam and Paul Roberts

4.1 Introduction

Historically, when sampling waterbodies, the practice of taking single discrete water samples has been utilised. Samples collected using these methods, however, only represent a discrete point in time and are not representative of the continuous exposure of organisms to the chemicals present. Factors such as fluctuations in water/tidal levels and effluent discharge intervals can play an important role in the levels of compounds present in a sample at any one time. An alternative way to sample, which results in a time weighted average concentration, is to use composite sampling technologies or passive samplers. The use of Polar Organic Chemical Integrative Sampling (POCIS) has been recently documented for the detection of various classes of compounds in the environment (Alvarez *et al.* (2004), Jones-Lepp *et al.* (2004) and Petty *et al.* (2004)) and these sampling devices are the subject of investigation in this study.

Using POCIS samplers, the validity and suitability of passive samplers to sample a broad range of unknown toxicants was investigated. The POCIS samplers were packed with a variety of different, commercially available, solid phase extraction (SPE) packing sorbents. Experiments were also conducted using POCIS samplers to establish the sampling rates of the devices under controlled conditions; in addition, the linearity of the uptake was determined. These data will enable mean waterborne concentrations over the sampling period to be derived.

In order to establish the preferred sorbent, solid phase extraction was carried out in spiked water samples. Historic methods are described elsewhere (Waldock and Thomas, 1999). These methods used a layered SPE system using ENV+ and C_8 in series to extract a broad range of organic contaminants. This system was tested alongside two recently developed polymeric sorbents; Oasis HLB and Strata-X. Results are shown in Table 4.1.

Table 4.1. Recoveries from different sorbent materials

Compound	Sorbent Recovery (n=5)/%		
	HLB	Strata-X	ENV+
Trimethoprim	47	55	0
Erythromycin	51	55	0
3,4,5-trichlorophenol	48	52	46
2,4,6-tribromophenol	30	32	1
Triclosan	55	58	51
Flutamide	58	63	61
Fluoranthene	86	90	83
p,p'-DDT	55	49	52
CB126	45	45	43
5-methylchrysene	62	55	50
5α-dihydrotestosterone (DHT)	90	106	33
Estrone	82	102	117
Tamoxifen	67	85	0
Dibenz[a,h]acridine	35	46	0
BDE100	65	64	57

These data show that both the HLB and Strata-X polymeric sorbents display greater retention for a wider range of compounds than the ENV+. For further developments using POCIS, Strata-X was used due to its generally higher recoveries of the compounds listed. Experiments to determine uptake showed that uptake was linear for all sequestered compounds after 15 days, indicating that this sorbent acts as an infinite sink for these contaminants during this time. Work conducted by Alvarez et al., 2004, indicate that these devices can act as an infinite sink for up to 56 days. Non-polar compounds, such as PAHs and DDT were not sequestered onto the POCIS sorbent since these are designed specifically for polar compounds. Semi-permeable membrane devices (SPMDs) are more appropriate for sampling less polar compounds.

Following this initial scoping work, POCIS packed with Strata-X have been used in the field to establish the oestrogenic and anti-androgenic potential of wastewater discharges from sewage treatment works (STW) and receiving waters. The results obtained using these devices were compared to those derived from the use of traditional, discrete 2.5 l water sampling. The results are shown in Table 4.2.

Some oestrogenic and anti-androgenic (receptor) activity was seen in all samples. At the sites nearest to the STW discharge, there is also acute toxicity to the yeast cells. Only when this toxicity is diluted out does the receptor activity become measurable.

A comparison of E2 and Flutamide equivalent concentrations derived from the POCIS extracts with respective activity measured in the 2.5 l grab samples indicates that, where receptor activity is relatively low, the results obtained using the POCIS and the grab samples are very similar (Table 4.2). Where the receptor activity is higher the POCIS sample results are much higher than the grab sample results. This may be due to the receptor activity in the grab samples being diluted out. If the acutely

Table 4.2. Oestrogenic and Anti-androgenic activity of STW discharge receiving waters on the River Ray, as equivalent concentrations of oestradiol (E2) and flutamide, respectively.

Distance from Rodbourne STW	3.5 km upstream	100 m downstream	1.7 km downstream	8.3 km downstream	Reference Site
Oestradiol Equivs. (ng l^{-1})	0.04	0.46†	0.8†	0.86	0.02
Oestradiol Equivs. (POCIS) (ng l^{-1})	0.03	2.62†	1.75†	0.94	0.03
Flutamide Equivs. (mg l^{-1})	0.12	0.02†	0.06†	0.15	<LOQ
Flutamide Equivs. (POCIS) (mg l^{-1})	0.03	1.15†	0.38†	0.27	0.03

† samples toxic to yeast at highest concentrations

toxic compounds are relatively non-polar, they will not accumulate on the POCIS, and so can be readily diluted out, revealing high receptor activity. In the grab samples, both acutely toxic and receptor active compounds are extracted, meaning that the amount of dilution required in order to cancel out the toxic effects are also diluting out the receptor activity. The difference in receptor activity may also be due to the integrative nature of the POCIS samplers as the grab samples may have been taken at a time of relatively low discharge, while the POCIS samplers take an integrated sample over approximately 30 days, including times of both high and low discharge.

The highest activity in terms of both E2 and flutamide equivalents is seen just below the sewage discharge. The activity observed in POCIS samples then declines with distance downstream from the treatment works. The lowest activities were recorded upstream of the STW, and at the reference site on the River Ock.

Further details of this work can be found on the EDCAT website at http://www.ceh.ac.uk/edcat.

5. Tri-butyl tin (TBT): a survey of imposex in dogwhelks (*Nucella lapillus*) in the UK and measurement of TBT in water and sediments in areas associated with high shipping activity

Authors: John Thain, Matthew Gubbins*, Myles O'Reilly* and Lynn McIlroy*

5.1 Introduction

TBT-specific biological effects monitoring was first established in the mid-1980s when the development of imposex in whelks and periwinkles was attributed to this compound. Imposex is the imposition of male sexual organs in female whelks and periwinkles and has been found to be a very sensitive indicator of TBT exposure. In severe cases of imposex it can lead to sterility in females and detrimental reproductive effects on individuals and populations. The data collected by 1987 using these two species provided the evidence for environmental damage and subsequently led to a UK ban on the use of TBT on small boats and in aquaculture. In 1989, the European Union imposed a similar ban (EU Council Directive (76/769/EEC)).

Over the past fifteen years, extensive surveys have been conducted to measure the prevalence of imposex in the UK. In 1992, in preparation for the 1993 North Sea Quality Status Report, FRS Marine Laboratory Aberdeen conducted a survey around the North Sea. To complement this study a further survey was conducted, using a similar sampling strategy, by laboratories in the countries around the Celtic Sea. In 1998, Defra funded FRS to conduct a further North Sea study with the added emphasis on including some 'hot spot' monitoring around ports and harbours.

An IMO International Convention on the Control of Harmful Anti-fouling Systems agreed at a Diplomatic Conference, in October 2001, to prohibit the application or re-application to ships of organostannic compounds as biocides in antifouling systems from 1 January 2003. This was implemented in the EU by Council Directive 2002/62/EC and, in the UK, approval for use of organostannic compounds acting as biocides in antifouling systems, granted under the Food and Environment Protection Act (FEPA) or the Control of Pesticides Regulations (COPR), have now been revoked.

In light of the EU ban and the revocation of approvals in the UK for the application of TBT, it was considered appropriate that a baseline survey of the effects and residual concentrations of TBT in UK waters be established. The data from such a survey would provide a baseline for further monitoring (trends) and provide data for the next State of Sea Report and also fulfil the UK's obligation to the OSPAR JAMP CEMP (Oslo Paris Commission, Joint Assessment Monitoring Programme, Co-ordinated Environmental Monitoring Programme) on issue 1.3. – to what extent do biological effects occur in the vicinity of major shipping routes, offshore installations, marinas and shipyards, etc.

The sampling programme included the measurement of imposex in shoreline dogwhelks (*Nucella lapillus*) and offshore whelks (*Buccinum undatum*), and the measurement of concentrations of TBT and DBT in water and sediments in areas of high shipping activity. The sampling was conducted in 2004.

5.2 Shoreline sampling and determination of imposex in dogwhelks (*Nucella lapillus*)

Shoreline populations of toothed adult *N. lapillus* were sampled by hand between spring low water and mid tide levels at sites used in the 1992 and 1998 surveys. 40-50 individuals were taken for analysis. The shell length of each animal was measured, and individuals were classified by their shell length according to observations by Moore (1936), ie juveniles (10–15 mm shell length), sub-adults (15–21 mm), and un-toothed adults (21–26 mm and 26–35 mm). At each of the juvenile and sub-adult survey sites, an attempt was made to obtain 20 individuals from each of the above size classes (and 40 toothed adults). Un-parasitised animals were extracted from their shells and viewed under a binocular light microscope fitted with a calibrated eyepiece graticule. *N. lapillus* were sexed and dissected to expose the reproductive organs and allow determination of imposex. The degree of imposex as measured by Vas Deferens Sequence Index (VDSI), was determined using international standard techniques (OSPAR, 2002).

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5.2.1 Determination of the Vas Deferens Sequence Index (VDSI)

The development of imposex in *N. lapillus* may be divided into seven stages, depending upon the developmental state of both the penis and *vas deferens* in the female (Gibbs *et al*., 1987). Stage 0 is identified where no signs of imposex can be seen. Stage 1 can be identified when the *vas deferens* begin at the site of the vulva with Stage 2 also showing a small penis behind the right eye tentacle. As imposex progresses, the *vas deferens* starts to develop from the penis (Stage 3) and will become continuous (Stage 4). Eventually, *vas deferens* tissue may proliferate over the opening of the vulva (Stage 5), rendering the female incapable of breeding since she can no longer release egg capsules. The trapped egg capsules form a solid mass within the capsule gland. In this final Stage (Stage 6), the capsule gland may eventually rupture, causing premature death of the female. Each of the seven Stages of imposex is known as a Vas Deferens Sequence (VDS) stage and calculation of the mean VDS for a group of females provides the Vas Deferens Sequence Index (VDSI) that may be used to compare the reproductive competency of different populations.

The VDS was determined for each female and the mean VDS calculated to provide an estimate of the VDSI of the population.

At all sites where *N. lapillus* was sampled around the British Isles, populations were found to have some incidence of imposex.

5.3 Assessment of imposex data against OSPAR assessment criteria

In order to aid environmental assessments, the Oslo and Paris Commission (OSPAR) have derived a set of biological effect assessment criteria for TBT, based on the development of imposex in gastropod species (OSPAR, 2004). For dogwhelks, these criteria are based on VDSI, and the values chosen relate to effects on the reproductive capability of females in the populations and the effects expected from exposure to TBT concentrations in water equivalent to the Environmental Assessment Criteria (EAC). The VDSI values used to discriminate 6 assessment classes (A-F) and the effects that these values relate to are given in Table 5.1. The VDSI data from the 2004 survey was assessed against the criteria presented in Table 5.1 and the results are shown in Figure 5.1.

5.4 Offshore sampling of whelks, sediment and water

Offshore populations of whelks were sampled using 2 m beam trawls towed for 10 minutes. OSPAR guidelines require >100 *Buccinum undatum* and >50 *Neptunia antiqua* to make a sample of either species. At no site were enough samples collected to meet the OSPAR guidelines.

Sediment and water samples were collected for the measurement of both DBT and TBT. The sampling took place during June and July of 2004 at offshore anchorages and shipping lanes as shown in Figures 5.2 and 5.3.

Anchorage sites were divided into grids that generally consisted of five stations ('corners of the anchorage and centre') where sediments were collected using a Day grab. The top 5 cm of sediment was collected from the grab using a stainless steel, hexane rinsed, scoop. Sediments were placed into 500 ml, hexane rinsed, Beatson jars, with hexane rinsed foil placed under the lid. The grab was cleaned between stations by thorough rinsing from the ships clean water supply. Sediment was extracted using standard analytical procedures and analysed using a gas chromatograph with a flame photometric detector (GC-FPD).

Sub-surface water samples (2.7 l) were collected in hexane rinsed, clean amber glass Winchester bottles, using a stainless steel water sampler. Samples were extracted using standard analytical procedures and analysed by GC-FPD.

Sediment concentrations of TBT were highest in samples from the Warp Anchorage off the Thames Estuary. Concentrations exceeding the limit of detection <0.001 mg kg^{-1} were also found in samples from anchorages off the Tees and Tyne estuaries, in Belfast Lough and within Milford Haven (Figure 5.2 and Table 5.2). Fewer sites had water concentrations of DBT and TBT above the limit of detection. Highest concentrations were found in Belfast Lough, although measurable concentrations were also found in Milford Haven, in the Mersey and off the Tees estuary.

Table 5.1. Oslo and Paris Commission biological effects assessment criteria for imposex in *N. lapillus*, based on VDSI (OSPAR, 2004).

Assessment class	*N. lapillus* VDSI	Effects and impacts
A	VDSI = <0.3	The level of imposex in the more sensitive gastropod species is close to zero (0 - ~30% of females have imposex) indicating exposure to TBT concentrations close to zero, which is the objective in the OSPAR strategy of hazardous substances.
B	VDSI = 0.3 - <2.0	The level of imposex in the more sensitive gastropod species (~30 – ~100 % of the females have imposex) indicates exposure to TBT concentrations below the EAC derived for TBT. eg adverse effects in the more sensitive taxa of the ecosystem caused by long-term exposure to TBT are predicted to be unlikely to occur.
C	VDSI = 2.0 - <4.0	The level of imposex in the more sensitive gastropod species indicates exposure to TBT concentrations higher than the EAC derived for TBT. eg there is a risk of adverse effects, such as reduced growth and recruitment, in the more sensitive taxa of the ecosystem caused by long-term exposure to TBT.
D	VDSI = 4.0 - 5.0	The reproductive capacity in the populations of the more sensitive gastropod species, such as *N. lapillus*, is affected as a result of the presence of sterile females, but some reproductively capable females remain. eg there is evidence of adverse effects, which can be directly associated with the exposure to TBT.
E	VDSI = > 5.0	Populations of the more sensitive gastropod species, such as *N. lapillus*, are unable to reproduce. The majority, if not all females within the population have been sterilized.
F	VDSI = -	The populations of the more sensitive gastropod species, such as *N. lapillus* and *Ocinebrina aciculata*, are absent/expired.

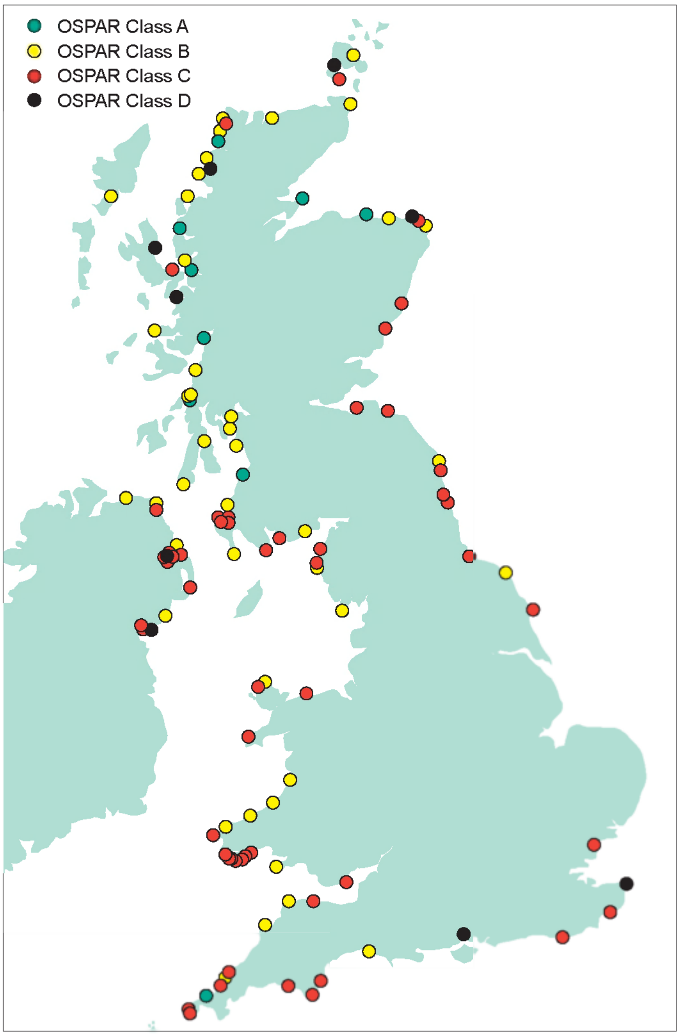

Figure 5.1. Assessment of 2004 VDSI data from adult dogwhelks (*N. lapillus*) sampled from sites around the UK. Data are presented in accordance with OSPAR assessment classes A-D.

Figure 5.2. Sediment sampling sites at offshore anchorages and shipping lanes. Black dots indicate sites at which measured levels of DBT or TBT were below the limit of detection (<0.001 mg kg^{-1}). Red dots indicate sites at which levels of DBT or TBT were above the limit of detection >0.001 mg kg^{-1}). See Table 5.2 for values.

Table 5.2. Sediment samples that had measurable concentrations of DBT or TBT (mg kg^{-1} dry weight) (see Figure 5.2 for site locations)

Site	Latitude	Longitude	Concentration (mg kg^{-1})	
Outer Sunk Anchorage	51° 54.76' N	1° 41.91' E	<0.001	0.010
Warp Anchorage (Thames)	51° 30.46' N	0° 53.59' E	0.016	0.586
Milford Haven	51° 41.90' N	4° 59.36' W	<0.001	0.024
Milford Haven	51° 41.94' N	5° 00.82' W	<0.002	0.027
Milford Haven	51° 41.90' N	5° 01.55' W	<0.002	0.130
Belfast Lough	54° 36.23' N	5° 55.11' W	0.024	0.159
Belfast Lough	54° 36.78' N	5° 54.59' W	0.035	0.344
Belfast Lough	54° 37.38' N	5° 53.48' W	0.021	0.088
Belfast Lough	54° 37.76' N	5° 52.97' W	<0.002	0.062
Belfast Lough	54° 38.06' N	5° 52.81' W	<0.002	0.033
Tees Estuary (Mouth, Buoy 12)	54° 41.08' N	1° 07.96' W	0.021	0.023
Tees Estuary (Dabholm Gut)	54° 36.86' N	1° 09.04' W	0.043	0.026
Tees Estuary (Harbour Masters)	54° 36.19' N	1° 09.60' W	0.060	0.098
Tyne Anchorage	55° 02.45' N	1° 02.47' W	<0.002	0.013
Tyne Anchorage	55° 01.63' N	1° 22.79' W	0.202	0.125

Figure 5.3. Water sample sites at offshore anchorages and shipping lanes. Black dots indicate sites at which measured concentrations of DBT or TBT were below the limit of detection. Red dots indicate sites at which levels of DBT or TBT were above the limit of detection. See Table 5.3 for values.

Table 5.3. Water samples that had measurable concentrations of DBT or TBT (ng l^{-1}). See Figure 5.3 for site locations

Site	Latitude	Longitude	DBT	TBT
Milford Haven	51° 41.90' N	04° 59.36' W	<1.40	4.18
Mersey Estuary	55° 27.23' N	03° 02.64' W	<2.03	8.44
Belfast Lough	54° 36.23' N	05° 55.11' W	7.79	<2.29
Belfast Lough	54° 37.76' N	05° 52.97' W	<1.98	9.40
Belfast Lough	54° 38.41' N	05° 52.37' W	<1.97	8.7
Tees Anchorage	54° 39.06' N	01° 03.31' W	6.4	<2.02
Tees Estuary (Harbour masters)	54° 36.19' N	01° 09.60' W	<1.76	7.79

5.5 Discussion

In the shoreline survey of imposex in dogwhelks all sites had female whelks with imposex. Applying the OSPAR assessment criteria to the data, of the 112 sites sampled: 9 were classified as A, the level of imposex was close to zero; 41 were classified as B, indicating exposure to TBT but with unlikely adverse effects; 52 were classified as C, a risk of adverse effects such as reduced growth and recruitment; and 10 sites were classified as D, presence of sterile females but some reproductively capable females remain. In general, all of the sites classified as C and D were associated with shipping activities so continued monitoring of these sites will be important in order to assess the effectiveness of the newly introduced IMO legislation. The lowest VDSI values were found at the greatest distance away from TBT inputs.

The shortage of whelks offshore, particularly at sites close to shipping lanes and anchorages limits the use of imposex in *B. undatum* and *N. antiqua* as a tool for monitoring the effects of exposure to TBT in these organisms.

Of the 34 sediment samples analysed, 15 had measurable concentrations of TBT above the limit of detection. The majority of these samples were muddy sediments taken from within estuaries associated with high shipping activity (Milford Haven, Belfast Lough and the Tees estuary). Of the anchorages sampled, two samples from the Tyne anchorage and one sample from each of the Thames Warp and Outer Sunk anchorages had measurable concentrations of TBT, but this in part may be due to the lack of sandy/muddy substrates frequently found at designated anchorages, eg Humber, Sunk and Tees.

Of the 32 water samples analysed only 5 had measureable concentrations of TBT above the detection limit and these were measured in water samples from Belfast Lough, Milford Haven and Tees estuary, enclosed sites associated with high shipping activity.

The results presented here suggest that monitoring of TBT in water and sediments is important in enclosed waters and may be particularly useful for assessing the effectiveness of the IMO ban on the use of TBT on large ships at these sites.

6. A proposed Marine Quality Index for the integration of chemistry, biological effects and biological community data obtained in the UK national monitoring programme

Authors: John Thain, Brian Miller and Mathew Service**

6.1 Introduction

The UK has in place a national marine monitoring programme now revised and known as the Clean Seas Environmental Monitoring Programme (CSEMP)), which incorporates chemical, biological effects and biological measurements at sites in offshore and coastal waters, and within estuaries. At present many of the techniques are deployed at the same sampling sites but the analysis and interpretation of the results are conducted by experts within each discipline rather than in an integrated way. A report by Defra entitled Safeguarding our Seas (Defra 2002) identified the need to develop better integration of marine environment monitoring and observation programmes as part of the ecosystem approach to management.

In order to develop this holistic integrated approach to monitoring, this investigation was funded by Defra with the purpose of developing a process of integrating data into a meaningful "ecosystem health index".

The development of an ecosystem health index requires consideration of conceptual as well as practical approaches. In this regard due consideration needs to be given to the tools currently available at each trophic level for the development of such an index (Table 6.1)

Table 6.1. offers an insight into the current difficulties in the development of indicators of ecosystem health, namely that there are currently some tools available, but that these operate at different levels in the ecosystem.

The conceptual task in developing an indicator of ecosystem health using the data from the UK monitoring programme is similarly difficult. This is because the data available for chemical contaminants, biology and biological effects only provide information on some levels of the ecosystem, as shown below, and uncertainties exist as to how this information should be conceptually linked.

It can be seen from Table 6.2 that there are currently no indicators of ecosystem health within the UK monitoring programme. Furthermore, chemical contaminant concentrations and biological effects data only provide information at a relatively low level in the ecosystem, at the sub-lethal or individual level. Recognising too that there are many definitions of ecosystem health the investigations in this study focussed on defining a health index.

It was recognized at the outset of this project that there currently exists no approach that integrates biological effects, benthic communities, and chemistry to produce an integrated health index. The purpose of this project was to attempt this integration, using data collected for the current UK monitoring programme for chemical contaminants, benthic biology and biological effects.

Table 6.1. Ecological relevance, natural variation and ease of standardisation of current tools.

Level	Ecological Relevance	Natural Variation	Ease of Standardisation
Ecosystem	No methods available		
Population/Community	High	High	Difficult
Individual (Bioassays)	Low to Intermediate	Not contaminant specific	Easy
Health (Biomarkers)	Low	High	Specific to some impacts
Biosensors	Low	High	Specific to some impacts

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Table 6.2. CSEMP data and ecosystem levels.

Level	Chemical contaminants	Biology	Biological effects
Ecosystem			
Community		Benthic infauna	
Population		Fish population data	
Individual		Fish disease	Scope for growth
Sub-lethal (Health)	TBT, PAHs, metals, CBs		Metallothionein, DNA adducts, EROD

6.2 Marine Quality Index

The aim in devising the Marine Quality Index (MQI), was to derive a quick and easy way of describing the ecosystem health at each CSEMP site in terms the quality of the waters, sediments and biota, as described by the chemical contaminant concentrations, biology and biological effects. To this end, a score ranging from 0 (good health) to 30 (bad health) and a colour (red, amber, green) was applied to each category, reflecting the quality or the degree of impact at the site (see Table 6.3).

Table 6.3. MQI, colour, status and score framework.

Colour	Health status	Score
Green	Good ecosystem health	0 – 10
Amber	Questionable health	11 – 20
Red	Bad ecosystem health	21 – 30

An assessment of the data types for biology, biological effects and chemistry quickly revealed that no single approach was applicable for the derivation of the respective indices. Therefore, each was derived independently. A full description on the process, details and calculations involved in deriving the index is lengthy and can be found in the Defra report for this project A1043 (www.defra.gov.uk).

The Marine Quality Index (MQI) score is constructed from individual scores derived for each of chemical contaminants, biological effects and benthic ecology.

A summary of the objectives and process used for each component was as follows:

6.3 Biology Index (BI)

The UK CSEMP assesses the biological status by using a series of selected parameters and indicators after testing their performance against quality assurance assessment criteria and OSPAR requirements. Benthic communities and biological effects are the main components of this assessment. Benthic invertebrates play an important role in the functioning of marine ecosystems and are well-established indicators in evaluations of environmental quality status. The assessment of benthic communities is important when assessing water and sediment quality because they indicate the condition of their environment and may uncover problems undetected or underestimated by other methods (chemistry and biological effects). Benthic community parameters provide indicators that will reflect directly upon responses at the community level, rather than at the population or even lower levels of biological organisation.

The EU Water Framework Directive (WFD) has defined ecological quality within five categories and includes some benthic parameters. The approach developed here considers such categories of ecological quality, but develops it further considering also other relevant benthic parameters and scoring systems to assess health status and can be integrated with chemistry and biological effects.

The biological component of the Marine Quality Index has been developed according to the following objectives:

- To develop a metric that allows the use of benthic parameters to contribute to the biological and ecological components of the overall assessment of environmental quality.
- To select the most suitable parameters from within the CSEMP programme to provide the benthic community assessment to formulate a Biology Index, and
- To develop a scoring system to integrate the benthic parameters and derive three main categories of health status using the red, amber and green colour coding system that correspond broadly to the five categories of ecological status established under the WFD.

The process involved:

- Assigning a weighting status to selected benthic parameters in terms of ecological significance.

6.4 Biological Effects Index (BEI)

Developing a simple metric of biological effects impacts within the scope of the current CSEMP is perhaps the most difficult aspect of this programme. Multivariate techniques to assess ecological changes are well established, and concentrations of chemicals have been related to benchmarks of animal health (eg death, leading to population and hence ecological change). However the linkage from sub-cellular responses in individual animals to whole organism level change is poorly defined for most CSEMP biological effects parameters and the interdependency of their responses are not established. Furthermore, the range of measurements made do not comprehensively cover all vital responses, survival, reproductive viability, growth and disease status for many species.

Bearing these limitations in mind, step one was to develop an index that might provide an overall weighted metric of biological effect and step two, to integrate this index with those for ecology and chemicals. The objectives for step one were:

- To develop a metric which will allow the use of all biological effects data irrespective of the technique, the species, or the level of organisation of the measured response.
- To develop a metric which can take account of the type of response (ie weighted such that higher level responses such as intersex have a greater influence on the index than lower level responses, such as sub-cellular effects).
- To develop a metric that will take account of varying numbers of responses and incomplete data sets.

The process involved:

- Ranking the significance of biological effects tests with the suite used in the CSEMP.
- Applying a categorisation scheme to evaluate the severity of effects at each monitoring station.
- Manipulating the data to mitigate the effects of missing information.
- Comparing the observed effects to the worst possible scenario to provide a measure of distance from worst case.
- Bringing the values into the same arithmetic range as those for the ecological and chemical indices to allow development of an overall integrated metric.

6.5 Chemical Contaminant Index (CCI)

The CSEMP specifies requirements for chemical contaminant monitoring in waters around the UK. Data have been collected at marine monitoring locations across the UK using these requirements, creating a database with several years of data. The CCI has been constructed and validated using the data collected within the CSEMP.

Within the chemical contaminants section of the MQI, the suite of contaminants used within CSEMP was limited to include measurements in fish tissue, sediment and shellfish tissue. Contaminants in water were not considered since data are relatively sparse and assessment of impact difficult. The selection of contaminants was narrowed to include those which have current and relevant environmental quality standards or limits available. These include heavy metals, polycyclic aromatic hydrocarbons (PAH), chlorobiphenyls (CB), and tri-butyl tin (TBT). There were a few contaminants that were not included in the MQI because the results are always low and of doubtful current relevance (ie. *p,p'*-DDE in shellfish). The index is designed to be flexible and can include newly monitored or high profile contaminants in the future, for example brominated flame retardants, if beneficial.

The chemical contaminants index (CCI) was developed according to the following objectives:

- To produce a scoring system for chemical contaminants that can be used in conjunction with ecological and biological effects scoring to give an overall assessment of ecosystem health at a site.
- To produce a scoring system for chemical contaminants based on CSEMP data that can be applied to varying and incomplete datasets in the same manner.
- To produce a scoring system based on relevant environmental quality standards and limits.
- To produce a scoring system that accounts for the relative toxicity of each contaminant group.

The process involved:

- Scoring each contaminant according to established quality standards
- Assigning a colour according to the associated risk
- Weighting groups of contaminants based on inherent hazard ranking
- Providing an indicative index score and assign a colour based on the associated risk
- Adjusting the total score to fall into the same arithmetic range as those for ecology and biological effects
- Expressing the results in terms of confidence of the assessment.

6.6 Development of a fully integrated Marine Quality Index (MQI) using CSEMP monitoring data for benthic biology, biological effects and chemistry

The above text outlines the development of the MQI health index for benthic biology, biological effects and chemistry. The scores for each individual index may be combined in different ways. One approach was to combine them into one overall Index by taking the mean of the three scores. However, a more powerful approach would be to use the colour coding to describe the site in terms of its chemical and biological quality, and in terms of the biological effects observed, as outlined below.

The combination of benthic ecology, biological effects and chemistry data for each monitoring site provides insights into the cause of environmental quality degradation for example:

- If the benthic Ecology is good but chemical and biological effects data indicate poor quality then there may be a problem

	Benthic Ecology	Biological Effects	Chemistry
Bad			x
Poor		x	
Good	x		

Rather than a simple tabular format, the colour coding system can be used to give a clear assessment of each site, as shown below.

Benthic Ecology	Biological effects	Chemistry

- The next example below indicates biological quality to be bad, but chemical and biological effects data indicate high quality, so the effect is likely to be due to physical effects or organic enrichment.

Benthic Ecology	Biological effects	Chemistry

- In this example, biology is bad and chemistry is bad, but the biological effects data indicate high quality, so the effect may be due to persistent and bioaccumulative compounds rather than toxicity.

Benthic Ecology	Biological effects	Chemistry

- Lastly, if biology and biological effects data are bad and chemistry is good then the effects are probably due to unmeasured contaminants.

Benthic Ecology	Biological effects	Chemistry

6.7 Limitations of the MQI

A full matrix of potential scenarios may be developed for each possible combination of values. In applying the Marine Quality Index to the UK CSEMP data set, the BI, BEI and CCI scores were calculated for the chemical contaminants, the benthic biology, and the biological effects for eighteen sites sampled from 1999- 2003. The results are as shown in Table 6.4.

The results show a further positive attribute of the MQI, and also an inherent flaw, where data are missing. The positive attribute is that where data for chemistry, biology and biological effects are present, a "shorthand method" of describing the quality of a site may be used. For example, the site at Tyne Hebburn (site 225) may be described as a "23, 15.2, 13.6" site in 2000, whereas the site at the Isle of Man for the same year may be described as a "6.2, 6.5, 5.0" site, indicating relative impacts.

The inherent flaw is that for several sites, data are missing, with the result that the MQI cannot be fully applied. It is hoped that this situation can be remedied in future.

Finally, there is a danger that the individual scores for chemistry, biology and biological effects could be applied in very inappropriate ways, for example in the simplistic application of a mean value. For example with the Clyde CMT 5 site 045 for 2003, the scores were 6.3 for biology (green), 21.4 for biological effects (red) and 22.5 for chemistry (red) with a mean of 16.7 (amber). In this instance the combined score would give an impression that the site is of questionable health, whereas two of the components, biological effects and chemistry are in the probable harm category. For this reason, mean values should not be used. Indeed, the power of the index is in the individual scores and in the ability to "drill down" through these to establish the specific contaminants of concern or the specific biological effects observed.

Table 6.4. MQI for eighteen CSEMP sites.

Location (CSEMP Station)	MQI for Biology / Effects / Chemistry ~ B / E / C														
	1999			2000			2001			2002			2003		
	B	E	C	B	E	C	B	E	C	B	E	C	B	E	C
Amble			5.0			10.0			5.0			10.0			10.0
Tyne Hebburn (225)	20.5		12.7	23.0	15.2	13.6	18.5	12.1	13.2	22.0		13.5		14.8	18.5
Tyne Ferry (235)	18.5		16.7	25.3		11.0	18.0		10.7	20.5		11.3			9.5
Off Tyne (245)	8.0	10.1	16.9	5.8	8.8	12.7		9.1	11.7	8.0		14.5		11.7	16.9
Off Tees (295)	4.3	8.7	5.2		6.4	10.0		7.3			4.2			6.2	
Firth (035)						7.8	12.5		8.4			4.4			5.3
Clyde CMT 5 (045)	7.3		20.1	7.3		19.3	7.3		16.2	7.3		13.1	6.3	21.4	22.5
Clyde CMT 7 (055)	3.8		24.1	3.5		24.1	5.3		18.7	4.3		14.6	5.3		22.1
Irvine Bay (070)	5.3		14.4	7.3		14.4	6.0		19.1	4.3		14.6	8.0	0.0	16.5
Loch Linnhe (076)	12.3		3.8	14.0		3.8	15.5		10.3	12.0		12.5	12.5		15.0
Liverpool Bay (715)		9.0	8.0	10.0	10.1	14.4		7.4	5.5		6.4	5.3		9.6	13.4
Isle of Man (805)	5.3	8.7	16.9	6.5	6.2	5.0	14.5	9.3	10.0		5.7	3.3	10.8	8.9	10.0
Belfast Lough (825)	3.3		12.5	3.8		6.3	6.3		1.3	6.6		2.3	6.0		7.5
Belfast Lough (845)	7.0		1.0	4.3		17.5	10.0		13.4	4.3		5.3	4.3	11.0	9.2
Cardigan Bay (655)				4.8	10.4	0.0		8.8	1.9		12.3	1.9		11.5	
Tees Phillips Buoy (325)	15.0			11.5		0.0	13.5			11.5					
Tees Bramlett's (305)	20.5			14.5			15.5			16.0					
Tees No 23 Buoy (315)	16.0			14.4			17.3			15.0					

6.8 Conclusions

The initial aims of this project were to:

- Review, assess and select suitable approaches and indicators for the assessment of the quality and health of the ecosystem, and
- Develop a conceptual model and translate this into an integrated index of health and quality, using data available from the UK CSEMP for chemical contaminants, benthic biology and biological effects
- While the project was successful in achieving these aims, several difficulties were identified, including:
- The complexities involved in trying to integrate results for many chemical contaminants in different matrices with a disparate range of biological effects, and with infaunal benthic community analysis.
- The relative lack of complete data sets available, even from an established monitoring programme such as the UK CSEMP.
- The relatively low levels at which chemical contaminants and biological effects act (ie at the sub-lethal or individual level), making it difficult to answer higher level questions about ecosystem health.
- The need to develop a conceptual model of how to address ecosystem health in terms of impacts at the community, population, individual and sub-lethal levels.
- The difficulties in trying to develop an index after the monitoring programme has been carried out and the results gathered, rather than starting the monitoring with the aim of developing an index in mind.

In consequence, the main project outcome was the development of:

- A Marine Quality Index, rather than a measure of ecosystem health
- The Marine Quality Index is seen as useful to:
  - Identify the quality of marine sediments and biota, in benthic communities and in biological effects.
  - Provide a handy shorthand method to describe the quality of a site as, for example, a 25, 15, 5 site in terms of the comparable chemical and biological quality, and the impacts as biological effects.

The project also included a considerable amount of work on developing an understanding of what is meant by "ecosystem health". Space limitations have prevented this work being included here. In summary, good ecosystem health in the marine environment has been taken to imply good water and sediment quality such that natural and biodiverse communities of species are supported, with normal structure, function and reproduction that are sustainable, resilient and maintained over time.

6.9 Recommendations for further work

The project was relatively short in duration, and time constraints prevented further development of the Marine Quality Index or its wider application to other data sets. For this reason, it is proposed that two areas for further work are:

- further development of the Marine Quality Index, particularly to assess whether relationships or linkages exist between the component parts, for example between specific contaminants and specific biological effects.
- wider application of the Marine Quality Index to other data sets.

Linked to this, there is a need to ensure that the UK CSEMP monitoring is carried out more fully, to provide a better data set for assessment of environmental quality using the Index in a second iteration.

It is further proposed that work is required to develop a conceptual model to link lower level effects to impacts higher up the ecosystem level, and to assess whether several lower level effects, for example at the individual or population level may be inferred as having effects at the community or ecosystem level.

Finally, if the first proposal made above produces useful results, it would be sensible to carry out specific monitoring as part of the UK CSEMP "with the aim in mind" of applying the Marine Quality Index to the data; this may mean that specific chemical contaminants found to cause specific biological effects are carried out in tandem to ensure that the Index may be more fully applied.

Healthy and biologically diverse

7. Advice on fishery implications of pipeline discharges

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7.1 Overview

This section gives a brief summary of activities carried out during 2005 in connection with the provision of advice on fishery implications of pipeline discharges.

Cefas appraisal of applications for pipeline discharges involves consideration of resources in the area, toxicity of the effluent, local hydrographic conditions and any standards set out in national policy or European Union (EU) Directives. This includes the impact of discharges on marine fauna, including fish nurseries and shellfish populations, and specific interactions of chemicals and the marine environment. One important issue in relation to sewage discharges is the microbiological contamination of bivalve mollusc shellfisheries and the associated human health concerns. The reduction of sewage contamination at source is the most effective way of reducing the health risk. It also reduces the burdens on the industry and increases acceptability of product to supermarkets.

7.2 Summary of pipeline discharge applications

During 2005 Cefas assessed applications for a total of 377 individual discharges; of these a total of 244 (65%) were assessed for their potential to impact on shellfish waters and/or production areas. Shellfish hygiene issues therefore continued to be the most common concerns addressed. Table 7.1 shows the types of discharge application commented on. The majority of applications were for discharge of domestic or combined domestic and industrial sewage, including storm and emergency sewage overflows.

Table 7.1. Numbers of applications of various types commented on within 2005.

Type of application	Number received
Sewage	359
Trade Effluent	18
Total applications received	377
(Shellfish related)	244

During periods of prolonged or heavy rainfall, surface water run-off will add to the domestic and industrial sewage substantially increasing the volume of wastewater entering the sewerage system. Although the sewerage networks and sewage treatment works are designed to store and treat multiples of the flows experienced in dry weather it is not feasible to treat all flows from these combined foul and surface water sewer systems. In pipe-full conditions storm overflows are required to allow a proportion of the untreated dilute sewage to discharge from the sewerage system. Intermittent discharges from storm tanks and combined sewer overflows (CSOs) may therefore by-pass the treatment works with the potential to impact on receiving waters.

Applications for 97 storm overflow discharges were received in 2005 of which 69 were identified as potentially impacting on shellfisheries. Where water company discharge improvements are identified to benefit shellfish waters, Environment Agency policy requirements restrict overflow operation to a maximum of ten spills per annum (in aggregation with other impacting storm discharges averaged over a ten year period). Previously this requirement had to be negotiated by Cefas on a case-by-case basis.

Cefas requested that the water companies provide an annual report of spills from each of these overflows for most of these and where necessary asked for clarification that discharges had been considered in aggregation with others impacting on the same fishery.

Emergency overflows only come into operation when there is a major failure at the sewage treatment works or pumping station. If this should occur, it may cause severe contamination of fisheries in the area. Applications for 61 emergency discharge applications were received, of these 37 had the potential to affect shellfisheries. In advising on such applications Cefas therefore requested urgent notification of emergency events to the local food authority so that appropriate action could be taken to protect public health.

A total of 57 applications were received for continuous discharges of secondary (biologically treated) effluent (39 affecting shellfisheries); these included upgrading of existing works and new treatment works and package plants. In addition 7 applications were received for tertiary treatment, of which 4 were for year-round UV disinfection aimed at reduction of microbiological contamination. All of these benefited shellfish areas.

7.3 Drivers for current pipeline discharge improvements

Most of the applications for discharge improvements were in order to meet the requirements of the EC Directive 91/271 concerning urban waste water treatment (European Communities, 1991a), EC Directive 79/923 on the quality required of shellfish waters (European Communities, 1979) and EC Directive 76/160 concerning the quality of bathing water (European Communities, 1976).

7.4 Shellfish waters objectives and discharge improvements

Human illness arising from the consumption of bivalve molluscan shellfish is a recognized problem. In the UK, the micro-organisms causing such illnesses are almost always the result of sewage contamination of the harvesting areas. The processing techniques, eg, purification in clean seawater, will not necessarily remove all of the illness-causing organisms. The processing requirements also have a significant practical impact on the business operation and costs for the producers.

The Shellfish Waters Directive (Council Directive 79/923/EEC) aims to protect and/or improve the quality of coastal and brackish water bodies in which shellfish live in order to contribute to the quality of edible shellfish products. Member States are required to adopt and implement programmes of pollution reduction and prevention to achieve guideline and imperative standards.

In 1999, ninety-five new shellfish waters were designated bringing the total to 119 in England in Wales. During 2004 a further review of shellfish water designations was undertaken and Cefas assisted Defra in the development of a new designation policy with the result that in November 2004 a further five new Shellfish Waters plus five extensions to existing waters were designated bringing the total to 124 waters in England and Wales. Designations confer protection to the shellfish production areas within them under the Shellfish Waters Directive imperative and guideline receiving water standards and associated design standards are incorporated into Environment Agency discharge consenting policy. The Government set a target for all commercially harvested shellfish beds to achieve a microbiological classification of at least Class B as categorised under EC Directive 91/492 (European Communities, 1991b), see Table 7.2 below.

Following this and for the first time, Shellfish Waters were included as a driver for investment in water company asset management programmes for 2000-2005 (AMP3). As a result, in 2001, a policy was developed by the Environment Agency for the consenting discharges to achieve the requirements of the shellfish waters directive and to ensure that schemes contributed to meeting Government targets. This policy drew significantly upon scientific advice and recommendations made by Cefas during the preceding years.

7.5 Impact of discharge improvements on bivalve molluscan shellfisheries

Between annual September classifications in 2000 to 2005 the percentage of Class A areas has decreased from 5.5% to 3.5% and the proportion of Class B areas increased from 66.4% to 86.5%. The proportion of Class C areas decreased from 24.7% to 8.1% and the number of prohibited areas decreased from 3.4% to 1.9% in the same period (Table 7.3). The increase in Class B areas and decrease in Class C and prohibited areas in this period, reflect water company investment targeting discharges to

Table 7.2. Classification categories under the Shellfish Hygiene Directive.

Class	Criteria	Requirements
A	<300 faecal coliforms or 230 *E. coli* per 100 g	Can be collected for direct human consumption
B	90% compliance with 6,000 faecal coliforms or 4,600 *E. coli* per 100 g	Must be purified or relayed to meet class A; may also be heat treated by an approved method
C	<60,000 faecal coliforms or <46,000 *E. coli* per 100 g	Must be relayed for at least 2 months to meet class A or B; may also be heat treated by an approved method
Prohibited	>60,000 faecal coliforms or >46,000 *E. coli* per 100 g	Commercial harvesting prohibited

Table 7.2(a). Percentage of Shellfish Production Areas at each Classification category in 2005.

Year	Class A	Class B	Class C	Prohibited
2005	3.5	86.5	8.1	1.9

tidal waters largely delivered since 1999 and in particular those improvements identified as benefiting shellfish waters in AMP3 to date. The decrease in the proportion of Class A areas is of concern however as Class A confers benefits of access to markets and premium market prices for the fishing industry. Despite significant improvements in water quality as a result of investment in discharge improvements in the last few years fewer than 4% of shellfish production areas in England and Wales achieve the 'Class A' standard which reflects water quality clean enough to allow marketing of shellfish direct for human consumption without further processing. To achieve Class A status for many fisheries it is likely that a combined programme of additional discharge improvements and/or measures to tackle diffuse pollution issues will be required in the future.

7.6 Drivers for future pipeline discharge improvements

This period marked the end of AMP3 scheme investment and the start of the AMP4 water companies investment programme 2005-2010 (AMP4, also known as PRO4). A number of schemes relating to further shellfishery benefits were costed under the initial Office of Water (OFWAT) Periodic Review 2004 (PRO4) planning. Following "Principal guidance from Ministers", the majority of these were removed from the National Environment Programme of planned improvements in AMP4. However it is the case that schemes targeted at meeting other drivers may also confer benefits to shellfish water quality indirectly.

The Environment Agency (EA) issued a proportion of water company AMP4 consent modifications in advance of scheme implementation and without consultation. This was to progress the regulatory requirement for compliance within shorter timescales than was achieved in AMP3. Cefas expressed concerns over the lack of consultation and recommended maintaining the consultation process for all schemes potentially impacting on shellfish waters. Cefas identified these relevant schemes and wrote to the EA with initial comments where relevant. The EA responded re-stating their position not to consult on these intended modifications, and responded to confirm that our comments had been noted. Cefas also requested copies of all modified consents from the EA to ensure that the discharges database is kept up to date.

Impacts of AMP3 schemes will now have an opportunity to become apparent, improvements in water quality in theory reflected by improvements in shellfish production classification status. The Environment Agency began to provide annual storm spill summaries of storm overflow events that impact on shellfisheries. This information will provide a management tool to assess success of scheme design, and to inform future investment targets. Notification of emergency overflow events to the EA and Local Food Authority required in AMP3 consents now provides the opportunity to actively manage shellfish harvesting to the benefit of protecting the public health of consumers.

7.7 General advice

In addition to applications for modifications to existing consents and new consents for sewage discharges, a variety of other advice relating to pipeline discharges was provided during the year.

This included advice given on the consent to discharge brine arising from solution mining, in connection with the creation of undersea Gas Caverns off Aldbrough (Yorkshire coast). Following review of the proposed scheme, Defra decided not to call in the application for determination by the Secretary of State. A steering group with Cefas representation has been established for monitoring of the discharge. Two further applications relating to solution mining of caverns for storage of gas under the seabed were commented on (Cleveland and Preesall).

General support was provided for water quality liaison with the Environment Agency regions. This has proved effective in championing the consideration of the marine environment and fisheries interests prior to any formal discharge consent application.

Cefas concerns over a trade discharge impacting on shellfish beds in the Menai Straits were raised with the EA. An EA investigation has been initiated and results of discharge and environmental sampling are awaited, along with discussions on possible resolution of the situation.

Consultation took place on a proposed First Time Rural Sewerage (FTRS) scheme in the Helford estuary catchment. Cefas advised the EA to pursue tertiary treatment as the resulting discharge impacts on a shellfish water and a shellfish production area. The industry is trying to regenerate the fishery in this area. Advice therefore also highlighted local shellfish industry concerns and recommended the involvement of all relevant stakeholders in discussions.

An initial assessment was made of a proposed trial using a novel chemical disinfectant. The advice to the EA recommended establishing the test organisms used to date, the actual efficacy of the product against human pathogens, and the potential for re-growth post disinfection. Cefas also highlighted earlier work on other hydrogen peroxide/acid based disinfectants.

Trade effluent applications for discharges to the River Tees, arising from ship dismantling were commented on. These interlink closely with related dredging and construction licensing activities that have been taking place at this "Environmental Reclamation and Recycling" facility and which Cefas has been consulted on. Comments therefore were made in a comprehensive way, considering the whole environmental scenario.

Other areas of advice included: advise in relation to improvements in Chichester Harbour; environmental consultants seeking advice on dissolved oxygen standards in tidal waters; advice to colleagues regarding the regulation of pipeline discharges and disinfectants used in finfish aquaculture and advice to EA Wales on shellfish processing plant effluent.

7.8 Water company appeal

Anglian Water Services successfully appealed against the EA consent requirements regarding reporting of emergency overflow and spill events for Grimsby Riby Street terminal pumping station (Humber Estuary) on the basis that the impacted Shellfish Production Area was not within a currently designated Shellfish Waters. Cefas advice had been given on the basis of protecting all shellfishery production areas not just those occurring in designated waters, given the potential impact of contamination on both public health and shellfish industry viability.

7.9 Supporting research

Support was provided in relation to a Defra/UK Water Industry Research/EA sponsored project that has been commissioned to look at the Impact of Intermittent Discharges on the Microbiological Quality of Shellfish. This is being undertaken by the Centre for Environment and Health at the University of Aberystwyth partnered by Cefas

7.10 Training and development

Staff attended a one-day Chartered Institute of Water and Environmental Management conference focused on the water industry periodic review in November 2005. Discussions focused on the improvement of procedures for investing in the water industry in a sustainable manner of benefit to the environment.

The 1st EMCO workshop "Analysis and removal of contaminants from wastewaters for the implementation of the Water Framework Directive (WFD)" in Dubrovnik focused on both contaminant analysis/emerging contaminants and new risk assessment tools.

An Environment Agency training course 'Water Quality Planning' was attended. The course covered the issues considered and tools used by the Agency when determining discharge consents and augmented the appreciation of factors involved in discharge consenting.

7.11 Database maintenance

All applications, consents and authorisations continue to be entered onto a database that contains details of all known discharges to saline waters in England and Wales. The database is being continually developed, provides unique intelligence and is a strategic tool used alongside other Cefas tools to underpin impact assessments and policy decisions in the marine environment. An integrated database is being developed, which combines several databases held within the shellfish hygiene group of Cefas. This integrated database will provide greater capacity to relate data and interrogate data, as well as improving quality control through auditing ability. Geographic Information Systems currently used independently, will also be incorporated, as this provides an invaluable tool in water quality work relating to discharges and the interaction with the marine environment.

8. Contaminants in marine mammals

Author: Robin Law

8.1 Introduction

Post-mortem examinations of harbour porpoises (*Phocoena phocoena*) regularly reveal heavy burdens of parasitic worms. Subsequent tissue analyses show varying levels of polychlorinated biphenyls (PCBs) accumulating in their blubber. Although a number of papers have documented geospatial and temporal changes in concentrations of chlorobiphenyls (CBs) and their detrimental effects on marine mammal health, as yet none have examined their possible role in determining nematode burdens in wild marine mammal populations. Using a dataset for porpoises stranded in the UK from 1989-2002, we found a significant, positive association between ΣCB levels (sum of 25 individual CB congeners) and nematode burdens, although the nature of the relationship was confounded with porpoise sex, age and cause of death. It was also apparent that the individuals with the highest infestations with nematodes did not have the highest ΣCB levels. Whilst PCB levels are important in determining nematode burdens, they are clearly not the sole determinants of nematode burdens in wild UK porpoise populations (Bull *et al*., 2006).

Law *et al*. (2006a) conducted analyses of two high-volume brominated flame retardants (BFRs), hexabromocyclododecane (HBCD) and tetrabromobisphenol-A (TBBP-A) in the blubber of harbour porpoises stranded or dying due to physical trauma (mostly incidental bycatch in fishing gear) in the UK during the period 1994–2003. Analysis was undertaken using LC/MS on a diastereoisomer-specific basis (Law *et al*., 2005; 2006b). Eighty-five samples were analysed for HBCD, and 68 of these for TBBP-A. TBBP-A was detected in only 18 samples and at low concentrations, from 6 to 35 µg kg^{-1} wet weight (Table 8.1). Although TBBP-A is a high-volume BFR, it is a reactive rather than an additive flame retardant (unlike HBCD) and so is bound within the matrix of the flame-retarded products, and so less likely to leach out. α-HBCD dominated over the other isomers and was detected in all samples analysed at concentrations ranging from 10 to 19,200 µg kg^{-1} wet weight. The maximum concentration was about double that seen in earlier UK studies. Investigation of possible time trends indicated a sharp increase in HBCD concentrations from about 2001 onwards (Figure 8.1), which was not confounded by porpoise length (as a surrogate for age), sex, nutritional status or location (Figure 8.2). Monte Carlo simulation was used to assess the significance of this increase, which proved to be significant at the 1% level (Figure 8.3). Further analyses of HBCD in porpoises will be undertaken during 2006.

Contaminant data deriving from the UK marine mammals strandings programme over the period 1993-2001 have been made available in a technical report (Law *et al*., 2006c). This supplements an earlier report which covered the period 1988–1992 (Law, 1994).

Table 8.1. Blubber concentrations of HBCD and TBBP-A (µg kg^{-1} wet weight).

Reference no.	Sex	Age (yrs)	Length (cm)	Area	Location	Date	%lipid	α-HBCD	β-HBCD	γ-HBCD	ΣHBCD	TBBP-A
SW1994/63	F	nk	127	East	Chantry Point, Orford, Suffolk	24/04/1994	88	64	< 5	< 5	64	17
SW1995/106B	M	< 1	104	Scotland	Portknockie Harbour, Grampian	24/08/1995	93	10	< 5	< 5	10	18
SW1995/142C	F	4	139	Scotland	Rosemarkie, Highland	22/12/1995	86	34	8	< 5	42	10
SW1996/52A	F	3	119	Scotland	off Shandwick, Highland	15/03/1996	92	29	< 5	< 5	29	13
SW1996/60B	M	1	107	Scotland	Cadboll Pier, Highland	20/03/1996	91	16	8	< 5	24	< 5
SW1996/84B	M	1	110	Scotland	Nigg Bay, Highland	06/05/1996	90	12	7	< 5	19	< 5

Table 8.1. continued: Blubber concentrations of HBCD and TBBP-A (μg kg-1 wet weight).

Reference no.	Sex	Age (yrs)	Length (cm)	Area	Location	Date	%lipid	α-HBCD	β-HBCD	γ-HBCD	∑HBCD	TBBP-A
SW1996/90B	M	< 1	122	Scotland	Balmedie, Grampian	21/05/1996	87	87	9	10	106	8
SW1996/126	F	nk	100	East	Whitby, North Yorkshire	25/07/1996	86	125	< 5	< 5	125	11
SW1996/139	F	nk	132	East	Redcar, Cleveland	27/08/1996	92	103	< 5	< 5	103	< 5
SW1996/175	F	nk	137	West	Nolton Haven, Pembrokeshire	09/12/1996	86	34	< 5	< 5	34	< 5
SW1997/143A	M	< 1	119	Scotland	Crovie, Grampian	13/09/1997	91	64	11	< 5	75	16
SW1997/174B	M	1	103	Scotland	bycatch off Loch Linnhe, Strathclyde	28/11/1997	90	434	< 12	< 5	434	< 5
SW1998/35B	F	12	161	Scotland	Troon, Strathclyde	28/02/1998	92	204	7	10	221	< 5
SW1998/58A	M	5	147	Scotland	bycatch, Grampian	26/0319/98	91	22	< 5	< 5	22	< 5
SW1998/105B	F	6	140	Scotland	Findochty, Grampian	10/06/1998	93	13	< 5	< 5	13	< 5
SW1998/155D	M	< 1	86	Scotland	Whitehills, Grampian	13/08/1998	91	62	< 5	< 5	62	35
SW1998/187	F	9	164	East	Salthouse, Norfolk	03/11/1998	85	59	23	< 5	82	8
SW1998/214A	M	< 1	100	Scotland	Portmahomack, Highland	28/12/1998	92	41	7	< 5	48	7
SW1999/17	F	< 1	101	East	Bridlington, East Yorkshire	28/01/1999	76	69	< 4	< 4	69	< 4
SW1999/25A	M	5	133	Scotland	bycatch St Andrews Bay, Fife	12/02/1999	88	209	7	< 5	216	< 5
SW1999/30A	F	5	140	Scotland	bycatch Sound of Bute, Strathclyde	17/02/1999	89	458	< 5	10	468	< 5
SW1999/63	M	nk	124	West	Pendine Sands, Carmarthenshire	29/03/1999	88	44	< 5	< 5	44	< 5
SW1999/72D	F	< 1	107	Scotland	Alturlie, Highland	09/04/1999	87	267	11	9	287	6
SW2000/16	M	nk	125	West	Port Eynon, Swansea	07/02/2000	91	56	< 5	< 5	56	< 5
SW2000/27	M	< 1	93	West	Llanon, Ceredigion	20/02/2000	85	218	12	< 5	240	< 5
SW2000/53	F	nk	115	West	Fishguard, Pembrokeshire	15/03/2000	86	42	< 5	< 5	42	< 5
SW2000/55	F	nk	134	East	Lowestoft, Suffolk	20/03/2000	83	39	< 4	< 4	39	< 4
SW2000/73	F	nk	152	East	off Sizewell, Suffolk	04/04/2000	84	41	< 5	< 5	41	< 5
SW2000/81	F	nk	150	East	Sea Palling, Norfolk	12/04/2000	84	103	< 5	< 5	103	< 5
SW2000/83A	F	1	106	Scotland	St Cyrus, Grampian	23/04/2000	92	192	< 5	9	201	6

Table 8.1. continued: Blubber concentrations of HBCD and TBBP-A (μg kg^{-1} wet weight).

Reference no.	Sex	Age (yrs)	Length (cm)	Area	Location	Date	%lipid	α-HBCD	β-HBCD	γ-HBCD	∑HBCD	TBBP-A
SW2000/103	M	nk	140	West	Aberystwyth, Ceredigon	09/06/2000	87	227	11	< 5	238	< 5
SW2000/131	M	nk	88	West	Rhos-on-Sea, Conwy	11/07/2000	83	203	13	< 4	216	13
SW2000/140	M	nk	140	East	off Bridlington, East Yorkshire	09/08/2000	86	221	16	13	250	13
SW2000/146(2)	M	nk	139	East	off Bridlington, East Yorkshire	29/08/2000	86	233	18	11	262	< 5
SW2000/150A	M	nk	125	East	off Bridlington, East Yorkshire	08/09/2000	56	125	< 3	6	131	9
SW2000/157	M	nk	112	West	Borth, Ceredigion	22/09/2000	88	93	< 5	11	104	< 5
SW2001/15B	F	4	148	Scotland	Lunan Bay, Tayside	18/01/2001	91	279	9	9	297	< 5
SW2001/21A	F	1	117	Scotland	Peterhead, Grampian	24/01/2001	90	346	13	9	368	< 5
SW2001/40	F	nk	105	West	Swansea Beach, Swansea	07/03/2001	88	875	24	13	912	< 5
SW2001/43A	F	< 1	114	Scotland	Carnoustie, Tayside	08/03/2001	87	103	14	8	125	< 5
SW2001/73A	M	< 1	112	Scotland	Aberdeen Beach, Grampian	30/03/2001	91	603	8	11	622	< 5
SW2001/79A	M	< 1	109	Scotland	Findhorn Bay, Highland	08/04/2001	92	176	< 5	9	185	< 5
SW2001/83A	M	1	118	Scotland	Crovie, Grampian	17/04/2001	89	141	8	< 5	149	< 5
SW2001/85D	M	< 1	110	Scotland	Balmedie Beach, Grampian	24/04/2001	85	10900	37	21	10958	< 5
SW2001/123	M	< 1	82	West	Pembrey, Carmarthenshire	13/06/2001	76	707	< 4	< 4	707	< 5
SW2001/140C	M	< 1	84	Scotland	St Andrews, Fife	07/07/2001	89	543	22	11	576	24
SW2001/183B	M	< 1	104	Scotland	Portknockie, Grampian	03/08/2001	92	2660	13	17	2689	12
SW2001/206D	F	4	137	Scotland	Burghead, Grampian	15/09/2001	88	201	< 5	< 5	201	< 5
SW2001/210	M	nk	110	West	Poppit Sands, Ceredigion	18/09/2001	90	1610	< 5	< 5	1610	< 5
SW2001/251	F	nk	108	West	Aberafon, Port Talbot,	09/11/2001	89	1490	15	< 5	1505	< 5
SW2001/253	M	nk	110	West	New Quay, Ceredigion	09/11/2001	89	222	< 5	< 5	222	< 5
SW2002/3	F	nk	122	West	Swansea Beach, Swansea	04/01/2002	88	1060	12	17	1089	< 5
SW2002/11A	M	nk	154	Scotland	Uiskentuie, Isle of Islay, Strathclyde	07/01/2002	91	496	< 5	< 5	496	na
SW2002/95	F	nk	156	West	Blackpool, Lancashire	07/02/2002	90	2690	12	< 5	2702	na

Table 8.1. continued: Blubber concentrations of HBCD and TBBP-A (μg kg^{-1} wet weight).

Reference no.	Sex	Age (yrs)	Length (cm)	Area	Location	Date	%lipid	α-HBCD	β-HBCD	γ-HBCD	∑HBCD	TBBP-A
SW2002/149B	M	1	116	Scotland	off South Sutor, Highland	02/04/2002	91	830	< 5	11	841	< 5
SW2002/169A	M	10	151	Scotland	Speybay, Grampian	16/04/2002	90	311	6	9	326	< 5
SW2002/170	M	nk	124	West	Porth y Post, Anglesey	17/04/2002	88	17600	9	< 5	17609	na
SW2002/194B	M	8	139	Scotland	Easter Skeld, Shetland	29/05/2002	90	172	11	9	191	< 5
SW2002/199D	F	4	155	Scotland	Easter Skeld, Shetland	01/06/2002	90	84	< 5	9	93	6
SW2002/214	F	nk	133	West	Tal-y-bont, Gwynedd	18/06/2002	89	15300	8	8	15315	na
SW2002/294E	F	nk	120	Scotland	Kames Bay, Isle of Bute, Strathclyde	15/08/2002	91	18400	18	9	18427	na
SW2002/308	F	nk	157	East	bycatch off Bridlington, East Yorkshire	02/09/2002	91	855	< 5	< 5	855	na
SW2002/309	M	nk	141	East	bycatch off Bridlington, East Yorkshire	02/09/2002	91	3420	< 5	13	3433	na
SW2002/311A	F	nk	150	Scotland	Blairmore, Dunoon, Strathclyde	06/09/2002	91	10400	29	13	10442	na
SW2002/321C	F	nk	160	Scotland	East links, North Berwick, Lothian	23/09/2002	90	330	26	< 20	356	< 20
SW2002/350C	M	nk	99	Scotland	Lundin links, Fife	11/11/2002	85	448	15	< 10	463	< 10
SW2002/351C	F	nk	166	Scotland	Otter ferry, Loch Fyne, Strathclyde	22/11/2002	82	7180	37	6	7223	na
SW2002/372A	M	nk	152	Scotland	St Andrews, Fife	11/12/2002	88	1340	< 5	14	1354	na
SW2002/372C	M	nk	104	Scotland	Balmedie, Grampian	13/12/2002	88	1030	15	9	1054	na
SW2003/159C	M	nk	104	Scotland	Braes, Isle of Skye, Highland	10/03/2003	92	17400	11	< 5	17411	na
SW2003/190	F	nk	151	West	Tywyn, Gwynedd	02/04/2003	87	143	< 5	12	155	< 5
SW2003/194	F	nk	139	East	Minsmere, Suffolk	05/04/2003	91	11440	< 5	< 5	11440	< 5
SW2003/220	M	nk	116	East	Hessle fore-shore, East Yorkshire	27/04/2003	85	7740	< 5	< 5	7740	< 5
SW2003/236	M	nk	110	East	Gibraltar Point, Lincolnshire	10/05/2003	83	8210	< 4	13	8223	< 5
SW2003/257C	M	nk	110	East	bycatch off Bridlington, East Yorkshire	08/06/2003	90	19200	< 5	8	19208	na

Table 8.1. continued: Blubber concentrations of HBCD and TBBP-A ($\mu g\ kg^{-1}$ wet weight).

Reference no.	Sex	Age (yrs)	Length (cm)	Area	Location	Date	%lipid	α-HBCD	β-HBCD	γ-HBCD	∑HBCD	TBBP-A
SW2003/260	M	nk	116	East	bycatch off Bridlington, East Yorkshire	11/06/2003	88	15900	20	18	15939	na
SW2003/271	F	nk	158	East	bycatch off Bridlington, East Yorkshire	25/06/2003	91	9460	< 5	< 5	9460	na
SW2003/274	F	nk	146	West	Tywyn, Gwynedd	27/06/2003	89	132	< 5	< 5	132	< 5
SW2003/296	F	nk	154	East	bycatch off Bridlington, East Yorkshire	23/07/2003	87	1070	< 5	8	1078	na
SW2003/312	F	nk	161	East	bycatch off Bridlington, East Yorkshire	05/08/2003	85	6570	54	21	6645	na
SW2003/334	M	nk	136	East	Minsmere cliffs, Suffolk	26/08/2003	85	2200	< 5	< 5	2200	< 5
SW2003/337	F	nk	155	West	Fishguard, Pembrokeshire	27/08/2003	88	1030	< 5	< 5	1030	< 5
SW2003/344A	F	nk	98	Scotland	Chanonry Point, Fortrose, Highland	09/09/2003	85	453	< 5	< 5	453	< 5
SW2003/353	F	nk	111	West	Morfa Dyffryn, Gwynedd	26/09/2003	88	9170	16	< 5	9186	< 5
SW2003/385	M	nk	153	East	Walton Backwaters, Essex	23/11/2003	88	8410	50	< 5	8460	< 5

nk: not known

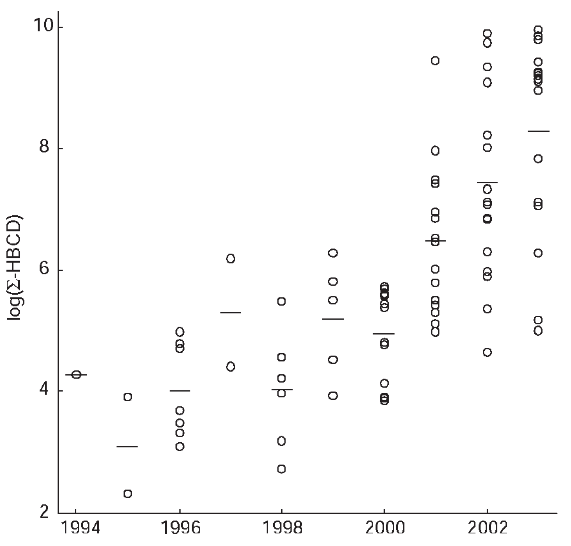


Figure 8.1. Plot of log(∑HBCD concentration) values against year of stranding or bycatch. Also marked are yearly means.

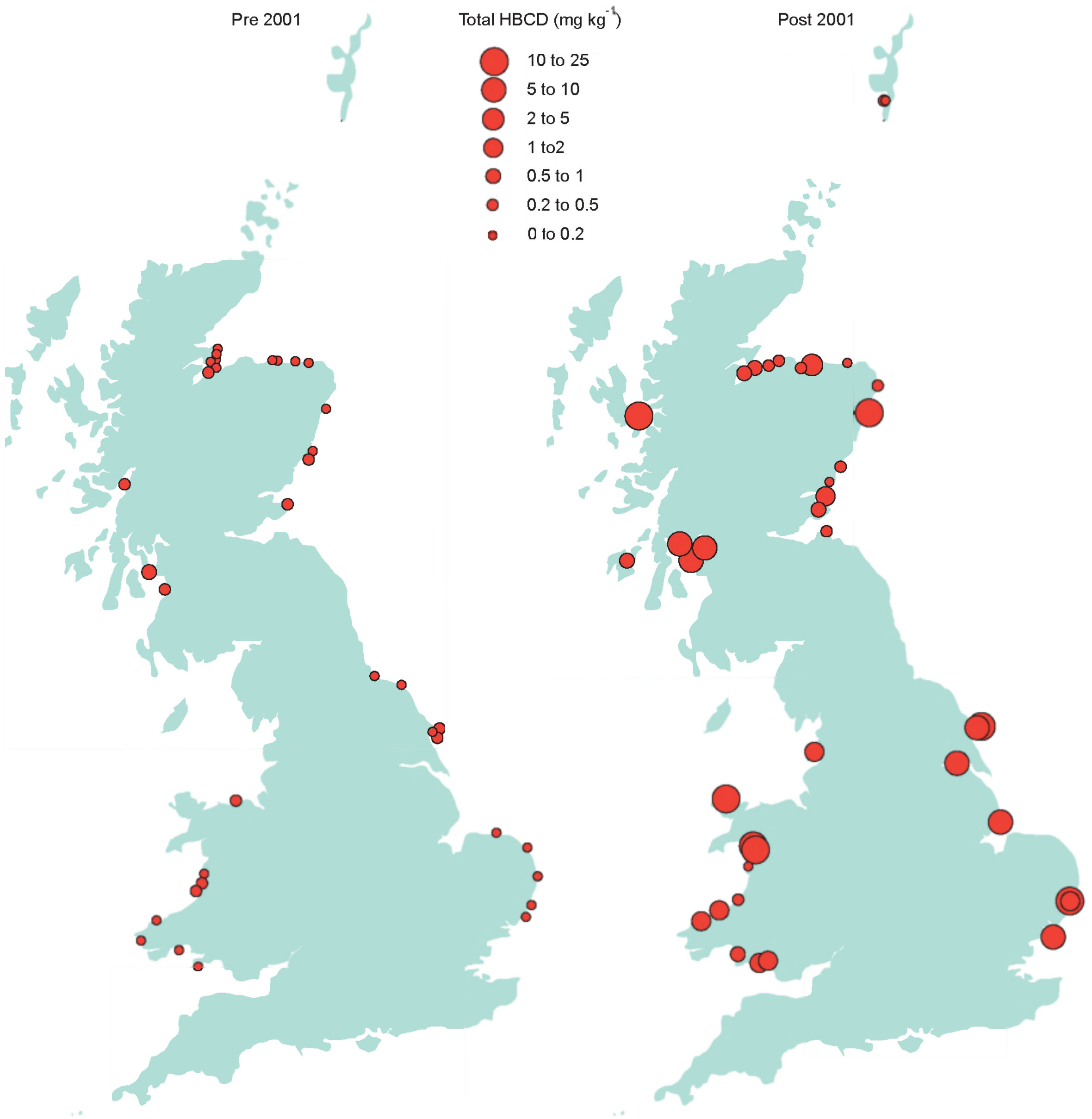
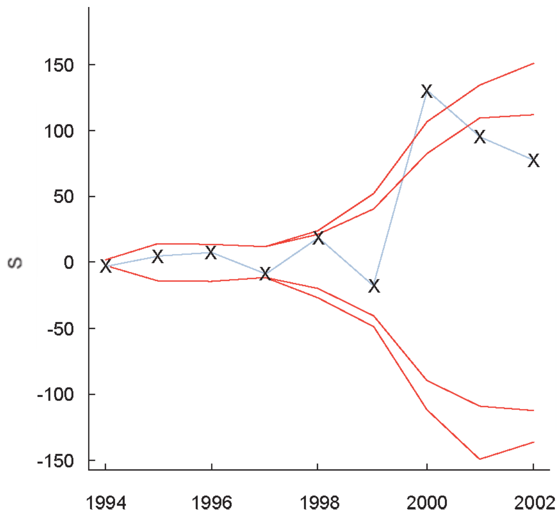


Figure 8.2. Spot diagrams indication ∑HBCD concentrations in porpoise blubber against location, up to the end of 2000 and from 2001 onward.

Figure 8.3. 95% and 99% envelope plot (red lines) for the statistic *S* comparing differences in ∑HBCD between successive years. The blue line is the observed value of *S*. Note that points on the plot represent comparisons between that year and the next year.



9. Health status of fish in the North Sea and Irish Sea 2004 and 2005 with a proposed site classification system based primarily on disease occurrence and prevalence

Authors: Stephen W. Feist, Brett Lyons, John Bignell and Grant D. Stentiford

9.1 Introduction

The use of fish diseases as a high level indicator of population health status has been used internationally in environmental monitoring programmes for more than two decades and provides a cost effective means to assess levels of environmental stress (Lang and Dethlefsen, 1996). Long-term monitoring has shown that changes in the prevalence of disease may be caused by a variety of factors, including contaminants (Hylland *et al*., 2006). The utility of this approach is recognised within the Oslo and Paris Commission (OSPAR) Co-ordinated Environmental Monitoring Programme (CEMP) (OSPAR, 1995), under which fish disease and liver histopathology are both accorded the highest CEMP ranking, to be conducted on a voluntary basis. Nationally as part of the Clean Seas Environmental Monitoring Programme (CSEMP), fish disease assessments form part of an integrated annual programme assessing biological and environmental factors in the seas around the UK for temporal trend monitoring and specific assessments at particular sites. On an international level, fish disease data have been used for environmental assessments within the framework of the North Sea Task Force and its Quality Status Report (North Sea Task Force, 1993), the OSPAR Quality Status Report 2000 (OSPAR Commission, 2000) and in the 3rd and 4th HELCOM assessments (HELCOM, 1996, 2002). Diverse activities in this field, including statistical analysis of long-term data on diseases of dab (*Limanda limanda*) in relation to contaminants and other environmental factors, have been undertaken by the ICES Working Group on Pathology and Diseases of Marine Organisms (WGPDMO) (Wosniok *et al*., 2000).

Externally visible fish diseases in a variety of fish species, including dab, flounder (*Platichthys flesus*) and cod (*Gadus morhua*) are applicable for general biological effects monitoring and are easily adaptable for other species such as whiting (*Merlangius merlangus*) and haddock (*Melanogrammus aeglefinus*). These include acute and healing ulcers, epidermal hyperplasia and papilloma, lymphocystis, hyperpigmentation and internally, the presence of macroscopic liver neoplasms. The presence of these conditions with information on the parasite burdens provides an overall indication of the health status of the fish populations. The fish disease monitoring programme also seeks to gather information on the health status of commercial fish species. In particular, conditions that may have implications for human health or those rendering the fish unsightly or unmarketable.

Within the OSPAR maritime area methodologies and diagnostic criteria involved in the monitoring of contaminant-specific macroscopic liver cancer and liver histopathology have largely been developed based on experience with benthic European flatfish species closely associated with contaminated sediments. Surveys have mainly involved dab and flounder, but can also be adapted to other flatfish species and also to benthic roundfish species (Stentiford *et al*., 2003; Feist *et al*., 2004). The dragonet (*Callionymus lyra*) that is present across a very wide geographic range of the eastern Atlantic from southern Iceland to the Canary Islands and Mediterranean regions has been identified as an alternative species for monitoring purposes by OSPAR. However, there is no published information on disease conditions affecting this species. Preliminary data from dragonet from the Irish Sea is provided in this chapter with comments on its potential use within monitoring programmes.

This chapter provides information on the disease status of a variety of fish species examined during 2004 and 2005. Building upon previous work, which demonstrated associations between disease status and geographic location and identified principle disease conditions which discriminate sites (Feist and Stentiford, 2005), we describe a new site classification system based primarily on disease occurrence and prevalence.

9.2 Materials and methods

Monitoring was undertaken as part of the integrated biological effects monitoring cruises that occur annually during the summer (June and July). In 2004 and 2005, nineteen sites were assessed for external fish disease and the presence of macroscopic liver nodules (For site locations see Figure 9.1). In 2004, eighteen sites and, in 2005, twenty-four sites, were sampled specifically for liver pathology.

Sampling protocols followed those established by ICES (Bucke *et al*., 1996) for external diseases. Target species were dab and cod (*Gadus morhua*) for offshore sites and flounder at inshore locations. Where sufficient numbers

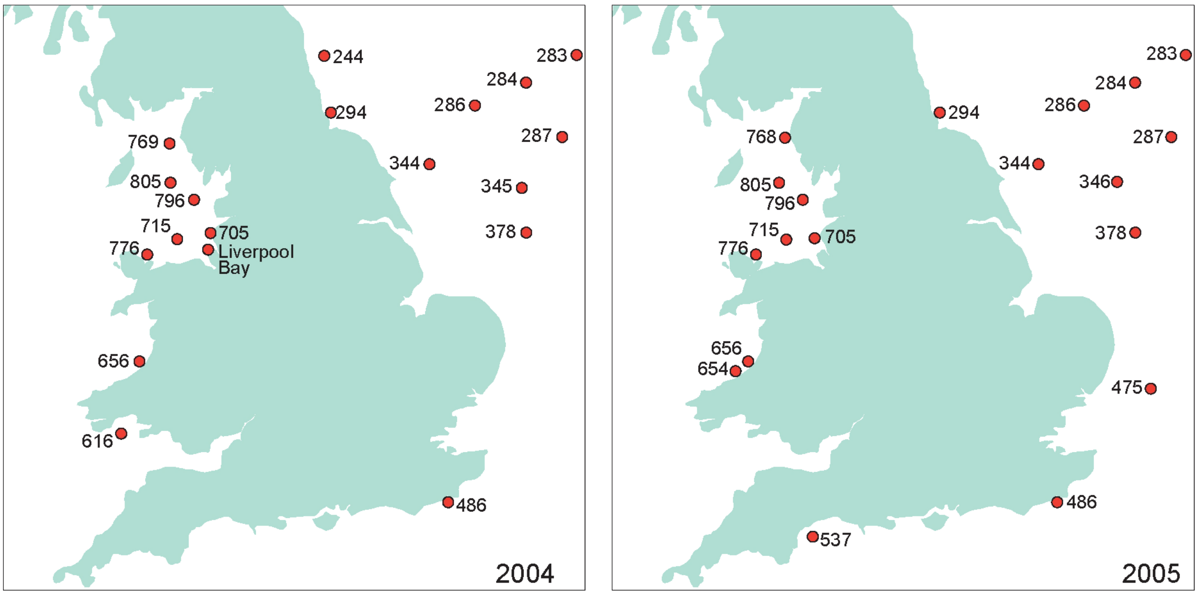


Figure 9.1. Locations sampled for external fish disease over 2004-2005.

of other species were caught a disease assessment was undertaken. Species sampled included plaice (*Pleuronectes platessa*) (fifty fish from Cardigan Bay were also sampled for analysis of liver histopathology during 2005, see section 9.3.5), haddock (*Melanogrammus aeglefinus*), whiting (*Merlangius merlangus*), pollack (*Pollachius pollachius*) and bass (*Dicentrarchus labrax*). In addition, during 2005 dragonets (*Callionymus lyra*) from Dundrum Bay (n=11), SE Isle of Man (n=21), St Bees (n=20) and inner Liverpool Bay (n=20) in the Irish Sea and at West Lundy in the western Bristol Channel (n=13) were examined for the presence of disease conditions and pathological changes in the liver, kidney, spleen and gonads.

From all dab examined for external disease that harboured liver nodules greater than 2 mm in diameter, a section of the liver incorporating the suspected tumour was taken for confirmatory histological analysis. Samples were fixed in neutral buffered formalin for between 24 and 48 hours and stored in 70% alcohol prior to further processing. Samples for other fish species requiring histological confirmation were treated similarly. In addition, standard sections of liver and gonad tissue were sampled from 50 dab greater than 20 cm in length for the assessment of pathological changes. At each site sampled, the first 20 fish were also sampled for other biomarkers. The otoliths were also removed from each of these fish for age assessment (data not reported here). Histological methods and diagnostic criteria followed those developed by ICES and were undertaken according to the quality assurance requirements required under the Biological Effects Quality Assurance in Monitoring (BEQUALM) programme (Feist *et al.*, 2004). Pathological changes to the liver are presented here under the broad categories of 1) non-specific inflammatory lesions, 2) non-neoplastic toxicopathic lesions, 3) foci of cellular alteration (FCA), 4) benign neoplasms and 5) malignant neoplasms. Fish displaying no liver pathology are reported as 'No Abnormalities Detected' (NAD).

Multivariate statistics using PRIMER software (Clarke and Warwick, 2001) were applied to the data since this approach provides increased sensitivity for the detection of differences in liver disease patterns between sites and between years compared to univariate analyses. Cluster analysis, Principle Component Analysis (PCA) and Multi-dimensional scaling (MDS) were employed to compare liver pathology data for all sites visited. The combination of these techniques allows for site similarity to be classified and importantly allows drivers for site similarity and difference to be identified. Nineteen and twenty-five sites were sampled in 2004 and 2005 respectively (Figures 9.2 and 9.3). The higher number of sites sampled for liver pathology in 2005 was due to the inclusion of new potential reference sampling sites in the English Channel.

9.3 Results

9.3.1 Dab diseases

Disease prevalence and severity data for dab sampled during 2004 and 2005 are presented in Tables 9.1 and 9.2. Data are shown according to size group. Overall disease levels remain broadly similar to those recorded in previous years with higher levels of cancer and other disease conditions in dab from Liverpool Bay in the Irish Sea and the Dogger Bank area in the North Sea (Cefas, 1998, 2000, 2005). Dab from Cardigan Bay and western Dogger Bank appear to show a trend of increasing prevalence of hyperpigmentation. This condition is prominent in North Sea dab and largely absent from dab sampled from Irish Sea sites. Disease levels in the Rye Bay, which has been used as a reference site, remain relatively low. As will be discussed later, the assignment of reference locations will likely need to be refined as data on the genetic similarity and demographics of fish populations together with their health status is incorporated in overall assessments.

Table 9.1. Summary catch data and disease prevalence in dab (*Limanda limanda*) by size category and disease severity on stations sampled during 2004.

CSEMP/ Area	Latitude/ Longitude	Size	M	F	LY	U	EP	HYP	LN	MLN	MA	X	ST	LP	AC	NM	GL
244	55° 19.234'N	15-19	85	36	4	2	1	3	0	0	2	0	23	0	0	14	0
Amble	01° 15.123'W	20-24	55	71	8	0	5	19	1	0	33	0	46	3	1	109	2
		25>	0	3	0	0	0	0	0	0	1	0	0	0	0	3	0
705	53° 28.255'N	15-19	13	6	0	7	1	0	0	0	1	0	1	8	0	0	2
Burbo Bight	03° 19.385'W	20-24	5	22	0	6	0	2	0	0	1	0	0	14	0	0	4
		25>	0	4	0	1	0	0	0	0	1	0	0	2	0	0	1
656	52° 17.6'N	15-19	143	57	4	15	6	31	0	0	0	0	16	38	0	0	0
Cardigan	04° 17.0'W	20-24	1	5	0	1	1	0	1	0	0	0	0	1	0	0	2
Bay Inner		25>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
616	51° 32.718'N	15-19	54	33	0	8	0	0	0	0	0	0	0	4	0	0	0
Carmarthen	04° 38.912'W	20-24	23	38	1	21	0	0	0	0	0	0	0	4	0	0	7
Bay		25>	1	38	1	5	0	0	3	0	0	0	0	5	1	2	3
287	54° 31.406'N	15-19	62	53	0	14	1	9	0	0	1	0	44	7	2	2	0
Dogger Central	02° 41.233'E	20-24	41	150	2	30	4	50	6	0	9	0	26	73	2	24	10
(Hospital Ground)		25>	0	39	0	4	1	21	7	0	1	0	3	22	3	13	0
284	55° 03.390'N	15-19	61	43	2	9	1	14	0	0	0	0	56	5	0	1	0
Dogger	02° 03.648'E	20-24	47	91	3	17	3	44	1	0	13	0	57	25	2	50	1
North		25>	0	61	1	4	2	17	3	0	4	0	17	14	2	21	1
283	55° 16.641'N	15-19	58	44	3	8	0	6	0	0	0	0	70	0	1	1	0
Dogger	02° 54.053'E	20-24	48	77	5	24	2	23	4	1	5	0	90	9	4	28	2
North East		25>	4	119	5	26	2	19	7	1	13	0	45	10	7	40	3
286	54° 45.317'N	15-19	68	43	3	4	2	14	0	0	1	0	80	1	0	4	0
Dogger	01° 18.886'E	20-24	38	97	0	10	5	51	3	0	8	0	82	16	2	62	2
West		25>	0	23	1	4	0	11	3	0	2	0	3	8	5	15	1
344	54° 15.00'N	15-19	73	47	5	3	2	10	0	0	0	0	28	0	1	19	0
Flamborough	00° 28.556'E	20-24	45	84	1	4	0	11	1	0	6	0	20	4	7	124	0
Off		25>	0	2	0	0	0	1	0	0	0	0	0	1	0	1	0
345	54° 03.238'N	15-19	53	61	4	1	1	9	0	0	0	0	78	4	2	5	0
Humber Off	01° 46.746'E	20-24	26	100	4	3	4	13	1	0	3	0	69	13	0	53	2
		25>	0	18	0	0	1	2	0	0	0	0	8	3	0	12	0
378	53° 32.566'N	15-19	52	65	0	2	3	4	0	0	0	0	4	2	0	2	0
Indefatigable	02° 05.583'E	20-24	14	110	0	6	2	26	1	0	1	0	0	26	4	17	3
Bank		25>	0	73	2	0	4	25	5	0	0	0	0	23	4	19	1
715	53° 28.252'N	15-19	62	33	0	6	1	0	0	0	0	0	9	4	3	0	1
Liverpool	03° 41.554'W	20-24	50	55	3	18	1	0	7	0	10	0	10	18	1	3	7
Bay		25>	0	47	1	12	2	0	8	4	3	0	1	16	0	1	3
Na	53° 23.449'N	15-19	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Liverpool	03° 34.372'W	20-24	2	32	1	7	1	1	0	0	2	0	0	18	0	0	3
Bay		25>	0	49	0	7	2	1	3	0	4	0	0	34	4	2	3
796	53° 55.430'N	15-19	62	49	0	11	2	0	0	0	0	0	27	24	0	0	1
Morecambe	03° 23.217'W	20-24	30	100	2	8	5	0	0	0	2	0	19	55	0	4	9
Bay Off		25>	0	59	0	7	2	0	3	0	1	0	4	36	0	8	4

Table 9.1. continued: Summary catch data and disease prevalence in dab (*Limanda limanda*) by size category and disease severity on stations sampled during 2004.

CSEMP/ Area	Latitude/ Longitude	Size	M	F	LY	U	EP	HYP	LN	MLN	MA	X	ST	LP	AC	NM	GL
776	53° 21.500'N	15-19	71	46	2	7	1	0	0	0	0	0	4	11	1	0	0
Red Wharf	04° 08.101'W	20-24	12	118	0	7	4	0	2	0	4	0	2	30	0	1	8
Bay		25>	0	33	2	4	2	0	0	0	2	0	0	15	0	0	0
486	50° 50.528'N	15-19	68	57	0	8	0	1	0	0	0	0	0	16	1	1	3
Rye Bay	00° 49.617'E	20-24	34	86	2	8	0	2	1	0	0	0	1	33	3	5	5
		25>	3	44	1	7	1	2	1	0	0	0	0	14	0	4	5
769	54° 32.960'N	15-19	168	55	1	21	3	0	3	0	0	0	135	35	0	0	1
St Bees	03° 50.456'W	20-24	9	37	0	6	2	0	3	0	1	0	18	20	2	0	0
		25>	0	6	0	1	0	0	2	0	0	0	2	1	0	0	0
805	54° 03.496'N	15-19	173	38	2	11	5	1	0	0	0	0	60	18	0	0	0
South East	03° 53.009'W	20-24	94	40	2	16	1	0	8	0	2	0	28	17	1	4	2
Isle of Man		25>	0	5	0	0	1	0	0	0	1	0	0	1	0	0	0
294	54° 45.833'N	15-19	47	59	7	4	0	0	0	0	1	0	28	1	2	2	0
Tees Bay	01° 08.161'W	20-24	38	106	10	1	1	7	2	0	9	0	44	7	1	110	1
		25>	2	9	0	0	0	2	0	0	1	0	2	0	1	9	1

Key: M = Male
F = Female
LY = Lymphocystis
U = Epidermal ulceration
EP = Epidermal papilloma
HYP = Hyperpigmentation
LN = Liver nodules
MLN = Multiple liver nodules
MA = Macrophage aggregates
X = x-cell disease
ST = *Stephanostomum* sp.
LP = *Lepeopthierius pectoralis*
AC = *Acanthochondria* sp.
NM = Nematodes
GL = *Glugea* sp.

Table 9.2. Summary catch data and disease prevalence in dab (*Limanda limanda*) by size category and disease severity on stations sampled during 2005.

CSEMP/ Area	Latitude/ Longitude	Size	M	F	LY	U	EP	HYP	LN	MLN	MA	X	ST	LP	AC	NM	GL
705	53° 28.220'N	15-19	163	81	0	30	3	3	0	0	5	0	5	68	1	1	10
Burbo Bight	03° 19.200'W	20-24	12	58	0	8	2	3	3	0	1	0	1	40	1	0	5
		25>	0	10	0	1	0	1	1	0	1	0	0	7	0	0	2
656	52° 18.497'N	15-19	117	70	2	4	1	13	3	0	2	0	9	26	1	1	2
Cardigan Bay	04° 15.357'W	20-24	3	8	0	2	0	2	1	0	1	0	0	2	0	1	1
Inner		25>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
654	52° 10.826'N	15-19	115	102	4	6	3	9	2	0	2	0	7	45	0	0	1
Cardigan Bay	04° 30.791'W	20-24	2	41	1	1	0	6	4	1	5	0	4	20	0	2	3
South		25>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
287	54° 31.190'N	15-19	41	62	2	10	2	7	0	0	0	0	29	4	1	0	0
Dogger	02° 41.290'E	20-24	45	95	2	31	2	42	7	0	4	0	23	47	3	20	13
Central		25>	1	17	1	4	2	6	4	0	3	0	1	12	0	4	0
283	55° 16.233'N	15-19	46	54	2	13	0	7	0	0	0	0	49	1	0	0	0
Dogger	02° 54.129'E	20-24	57	76	6	36	4	41	8	4	6	0	71	32	2	33	5
North East		25>	6	61	0	14	3	30	7	1	2	0	14	22	1	25	1
284	55° 03.509'N	15-19	77	28	2	16	1	39	0	0	0	0	49	9	1	0	0
Dogger North	02° 03.528'E	20-24	57	79	2	21	6	55	7	3	12	0	62	37	2	48	0
		25>	2	56	4	6	1	25	5	0	2	0	10	22	2	28	0
286	54° 47.610'N	15-19	73	31	3	4	1	35	0	0	1	1	65	3	0	1	0
Dogger West	01° 16.900'E	20-24	41	102	4	10	5	67	8	1	9	0	83	20	1	44	4
		25>	1	9	0	2	0	8	2	0	0	0	3	3	0	7	0
344	54°15.020'N	15-19	82	26	6	3	0	8	0	0	1	0	16	0	0	7	0
Flamborough	00° 28.790'E	20-24	50	90	5	3	0	32	5	0	9	0	17	4	1	125	0
Off		25>	1	9	1	0	1	3	0	0	1	0	0	2	0	10	0
346	54° 04.877'N	15-19	57	45	4	2	0	4	0	0	0	0	48	0	0	1	0
Humber Off	01° 48.800'E	20-24	36	106	6	2	5	11	5	1	3	0	45	10	8	72	2
		25>	0	30	0	3	0	5	0	0	0	0	5	3	2	22	0
378	53° 34.210'N	15-19	27	77	0	2	1	2	0	0	0	0	4	3	1	0	0
Indefatigable	02° 05.330'E	20-24	12	114	1	7	4	18	4	0	0	0	1	22	1	34	1
Bank		25>	0	86	2	3	3	25	13	5	1	0	1	26	3	37	1
715	53° 25.450'N	15-19	135	73	3	16	4	5	0	0	1	0	3	24	0	0	1
Liverpool Bay	03° 41.770'W	20-24	58	68	0	21	3	3	6	2	6	0	2	44	0	1	20
		25>	0	35	0	1	2	1	5	1	1	0	1	20	0	1	5
537	50° 33.740'N	15-19	74	34	0	0	1	2	0	0	0	0	0	0	0	2	2
Lyme Bay	02° 45.350'W	20-24	12	31	0	4	0	1	0	0	0	0	0	1	1	4	1
		25>	4	24	0	1	0	1	1	1	0	0	0	2	0	2	1
796	53° 54.184'N	15-19	122	84	0	12	1	0	0	0	0	0	5	51	0	0	2
Morecambe	03° 24.181'W	20-24	37	197	0	23	5	0	4	0	1	0	4	128	0	3	14
Bay		25>	0	81	0	10	2	0	3	0	0	0	2	60	0	0	10
776	53° 21.925'N	15-19	121	89	3	24	2	0	0	0	0	0	6	48	0	0	0
Red Wharf	04° 10.277'W	20-24	21	104	1	5	3	0	3	0	0	0	0	51	0	3	3
Bay		25>	1	36	0	2	0	0	0	0	0	0	0	20	1	0	1

Table 9.2. continued: Summary catch data and disease prevalence in dab (*Limanda limanda*) by size category and disease severity on stations sampled during 2005.

CSEMP/ Area	Latitude/ Longitude	Size	M	F	LY	U	EP	HYP	LN	MLN	MA	X	ST	LP	AC	NM	GL
486	50° 45.840'N	15-19	89	24	0	1	2	0	0	0	0	0	2	3	2	1	0
Rye Bay	00° 44.760'E	20-24	32	95	0	9	1	1	0	0	0	0	0	27	5	9	11
		25>	0	25	0	1	0	1	0	0	0	0	0	9	0	2	6
768	54° 32.850'N	15-19	169	56	4	26	2	0	0	0	1	0	115	42	0	0	1
St Bees	03° 50.070'W	20-24	20	47	1	8	1	0	0	1	4	0	20	30	1	0	1
		25>	0	7	0	1	0	0	0	0	0	0	0	6	0	1	0
805	54° 03.321'N	15-19	161	47	3	16	1	0	0	0	0	0	55	21	0	0	0
South East	03° 49.484'W	20-24	43	28	1	12	2	0	9	1	3	0	20	23	0	1	1
Isle Of Man		25>	0	4	0	0	0	0	2	1	0	0	0	3	0	0	0
294	54° 46.042'N	15-19	28	72	0	2	0	3	0	0	0	0	7	0	0	0	0
Tees Bay	01° 08.249'W	20-24	44	100	2	0	0	4	1	0	3	0	21	2	0	140	0
		25>	1	11	0	1	0	1	0	0	0	0	0	0	0	12	0
475	52° 02.470'N	15-19	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Thames	02° 06.740'E	20-24	11	16	0	1	0	1	0	0	0	0	0	1	0	7	3
Gabbard		25>	0	18	0	3	0	2	1	0	1	0	0	1	0	10	3

Key: M = Male
F = Female
LY = Lymphocystis
U = Epidermal ulceration
EP = Epidermal papilloma
HYP = Hyperpigmentation
LN = Liver nodules
MLN = Multiple liver nodules
MA = Macrophage aggregates
X = x-cell disease
ST = *Stephanostomum* sp.
LP = *Lepeopthierius pectoralis*
AC = *Acanthochondria* sp.
NM = Nematodes
GL = *Glugea* sp.

9.3.2 Assessment of dab liver pathology

The MDS ordination plot for 2004 clearly discriminates three major groups of sampling locations based on liver pathology in dab, each contained within 85% similarity contours. One group comprising stations on Dogger Bank and in Liverpool Bay and Cardigan Bays; a second group consisting of other North and Irish Sea stations, and a third group containing Red Wharf Bay, Morecambe Bay, Rye Bay and the station in Tees Bay in the North Sea (Figure 9.2(a)). The discrimination pattern is driven by the highest prevalences of preneoplastic and neoplastic lesions (benign and malignant liver tumours) in fish from sites located in the upper left quadrant of the plot and a lower prevalence in fish from sites towards the bottom right quadrant (Figure 9.2(b)). The MDS ordination plot for the comparative data collected during 2005 once again depicts the clear discrimination between these three groups. Although there is an overlap between two of them, each are contained within 80% similarity contours (Figure 9.3(a)). The overall distribution pattern of the sites is similar to that seen in the 2004 MDS plot with one group consisting of Dogger Bank sites, Cardigan Bay and a relative outlier, Gabbard. A second group, which overlaps with the first contains a similar mix of North Sea and Irish Sea stations to that produced from the 2004 liver pathology data. The final group, to the lower left of the ordination plot again contains Red Wharf Bay and Rye Bay. It should be noted however, that in 2004, fewer locations were sampled than in 2005 making a direct comparison between the years impossible using this approach. For year to year comparisons using PRIMER it is important to use the same sampling sites in each of the years. For example, fish captured from the Tees Bay site displayed liver conditions that placed them in the lower right quadrant of the MDS plot for 2004, with the site position moving to a more central relative position during 2005. The requirement for year to year trend analysis of such multivariate data highlights the necessity to sample sites consistently in consecutive years. Nevertheless, the bubble plots for 2005 show that the main drivers for the site to site discrimination remain consistent between years (Figure 9.3(a) and (b)).

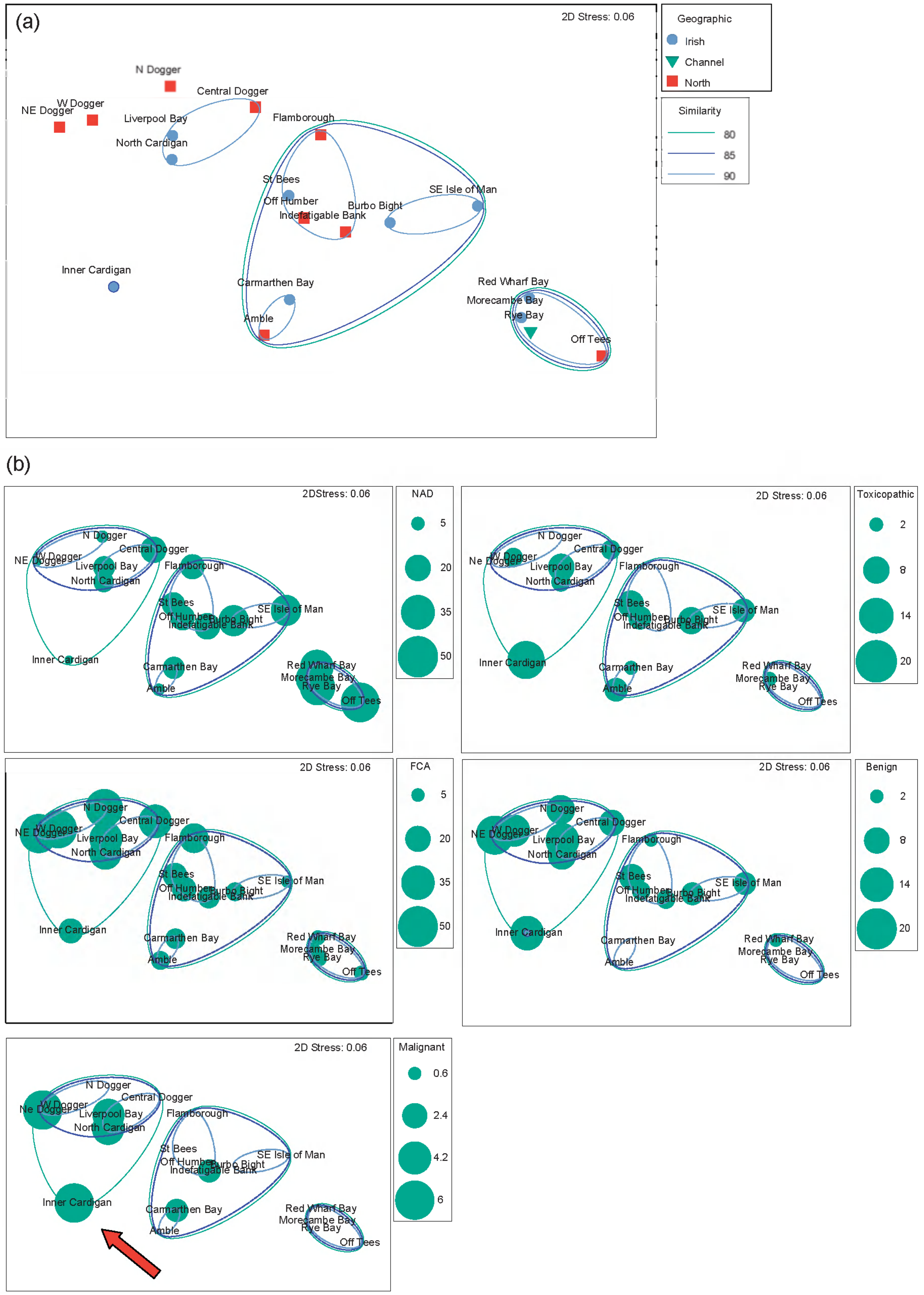


Figure 9.2. (a) Site discrimination based on liver pathology (2004 data), (b) drivers for site discrimination based on liver pathology (2004 data).

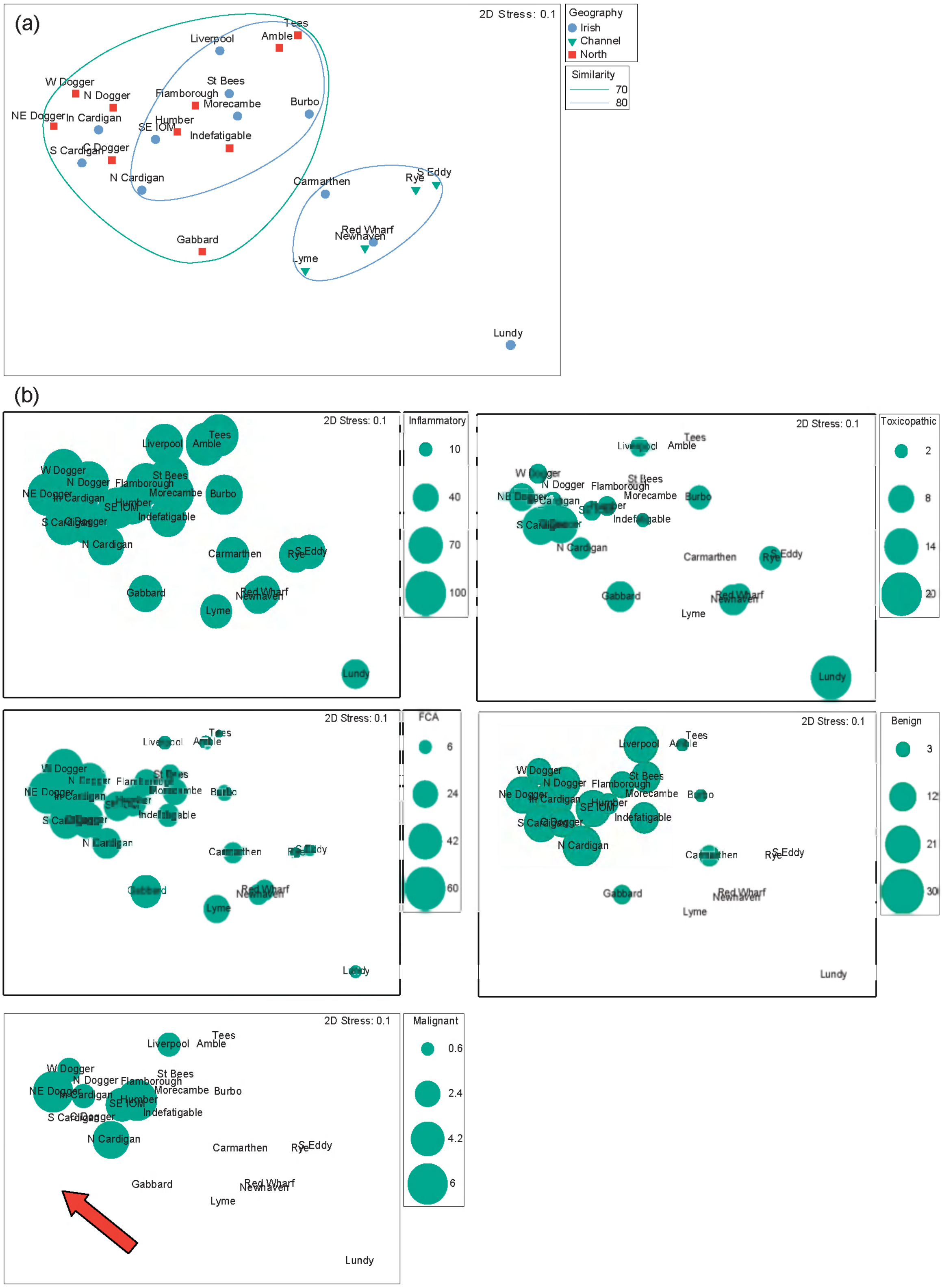


Figure 9.3. (a) Site discrimination based on liver pathology (2005 data), (b) drivers for site discrimination based on liver pathology (2005 data).

9.3.3 Site classification using disease levels and liver histopathology

Cluster analysis of the 2005 data allowed the allocation of individual sites to one of three site types (here termed Type A, B and C) (Figure 9.4). As for MDS, sites cluster more closely to one another when the array of pathologies observed in the liver of fish from those sites was most similar. Conversely, sites cluster least closely when fish from those sites show the largest differences in liver condition. Using this semi-quantitative approach to site classification, it is possible to show that sites do not necessarily group most closely to one another based upon geographical relatedness but rather that sites that are distant to one another can contain fish which show a very similar array of liver diseases. Conversely, geographically related sites may contain fish that do not show similar liver pathologies. Using this approach, we have allocated sites to either Type A, B or C, with C types containing fish with the highest prevalence of 'top level' neoplastic lesions and A types with the lowest prevalences of 'top level' neoplastic lesions. Using the same approach on data obtained from 2004 cruise, it was found that the majority of sites fell within the same site types (Figure 9.5). Furthermore, using these new designations of site type, it was then possible to cross-correlate other specific features recorded in fish captured from these site types (e.g. the external disease status of fish captured from site types A, B and C) and a working definition of each site type was constructed (see Figure 9.6). Type A sites are those in which dab have low levels of ICES-defined external diseases and low prevalence of liver pathology (including liver cancer), in type B sites dab exhibit increased disease levels of up to 20% and few cases of malignant cancers. Dab from type C sites show elevated prevalences of several external disease conditions, few fish without liver pathology, relatively high prevalences of preneoplastic liver pathology and increased numbers of fish with benign and malignant tumours. Analysis of site type data between years will allow for overall statements of improvement, decline or stasis in fish population health in given locations.

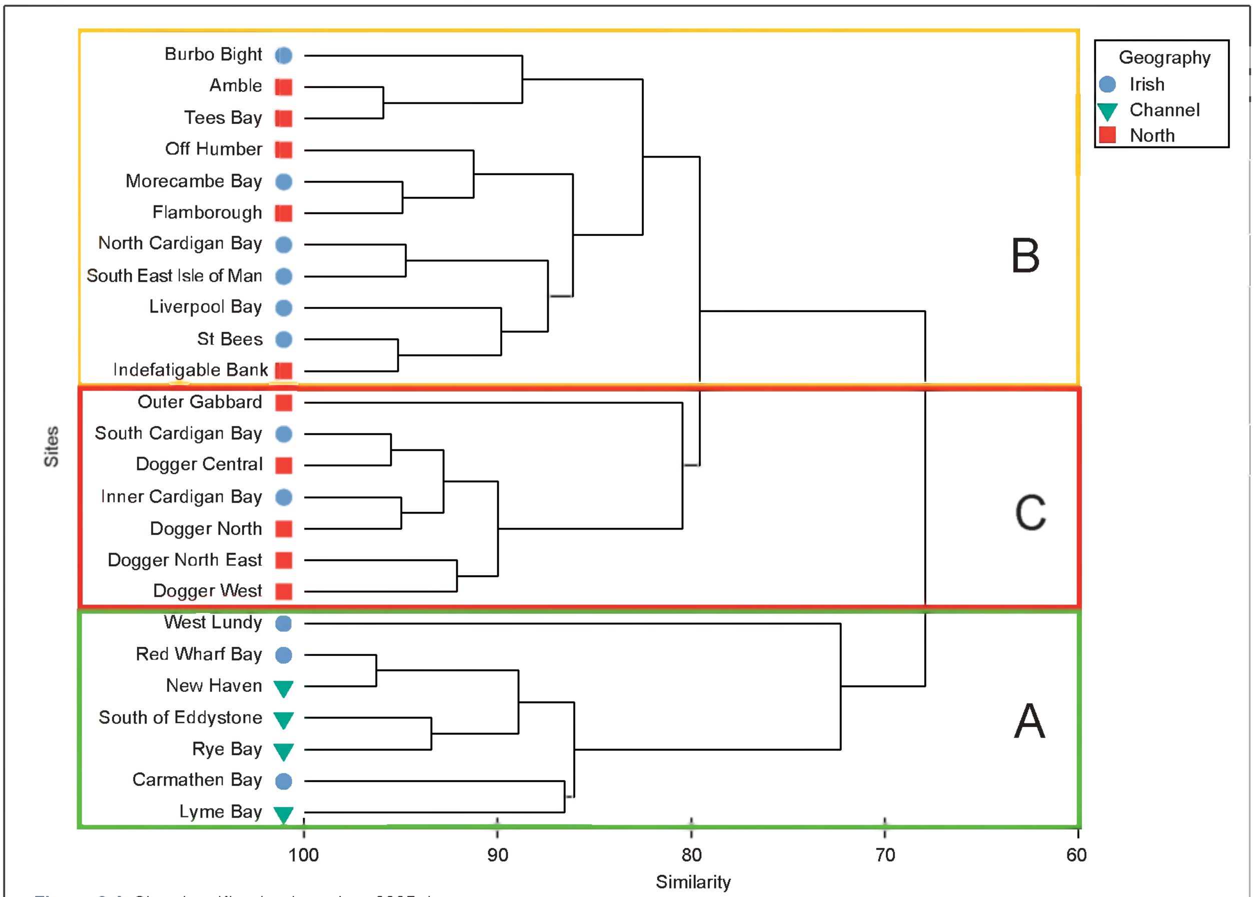


Figure 9.4. Site classification based on 2005 data.

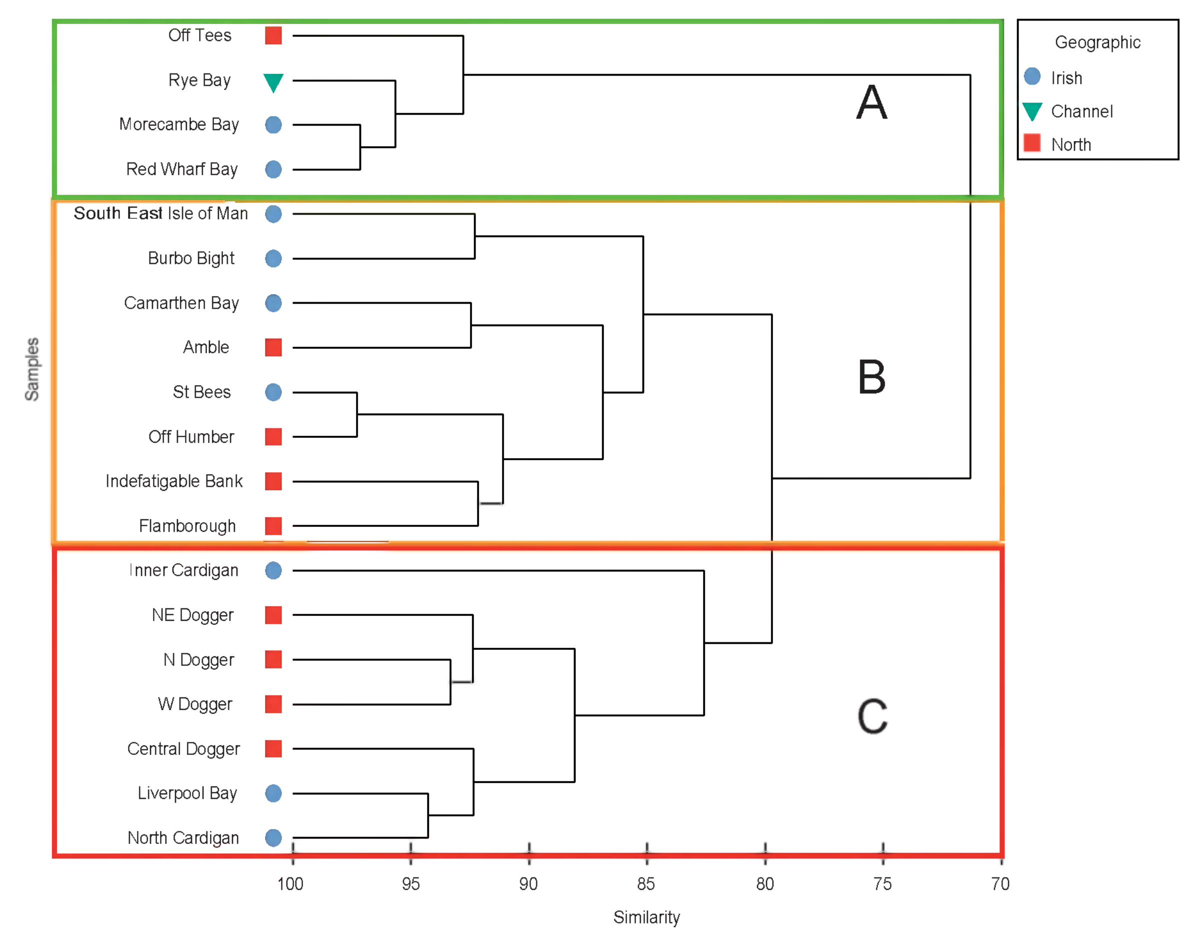
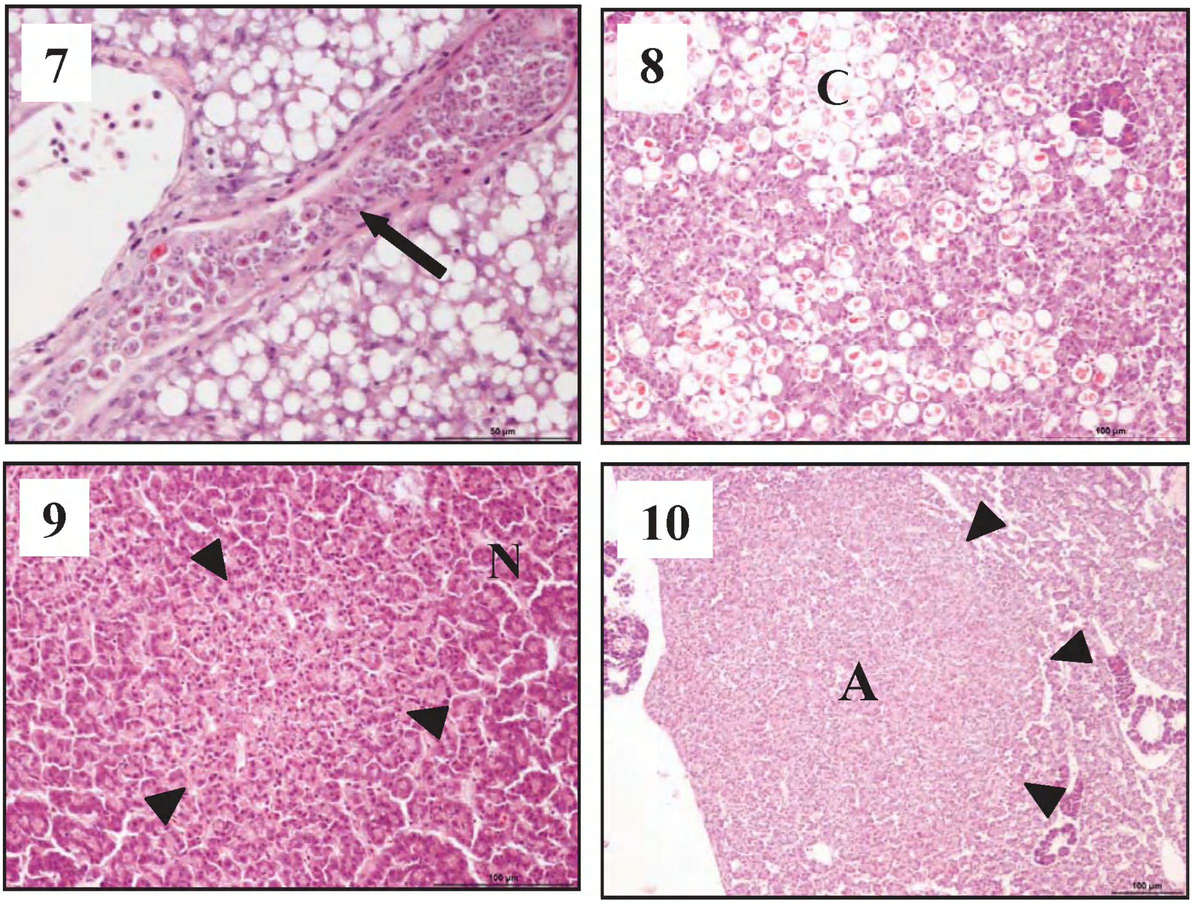


Figure 9.5. Site classification based on 2004 data.

Type A	Type B	Type C
• Generally low levels of ICES external diseases and almost complete absence of skin hyperpigmentation. • Approximately 30% of fish with no indication of BEQUALM liver pathology categories. • Low prevalence (<5%) of toxicopathic lesions and approximately 50% prevalence of inflammatory lesions (according to BEQUALM). • Low prevalence of FCA (<15%), benign tumour (<5%) and malignant tumour (0%) according to BEQUALM.	• Appearance of higher prevalence of ICES external diseases (incl. Lymphocystis and skin hyperpigmentation, the latter up to 20% at North Sea sites). • Between 10 and 20% of fish with no indication of BEQUALM liver pathology categories. • Low prevalence (generally <5%) of toxicopathic lesions but an elevated prevalence of inflammatory lesions (up to 90%) compared to Type A sites. • Prevalence of FCA can exceed 15% with mean benign tumour prevalence >10%. Appearance of malignant tumours at low prevalence.	• Elevated prevalence of several ICES external diseases (incl. Ulceration, parasites and skin hyperpigmentation, the latter up to 50% at some Dogger sites). • Low prevalence (<10%) of fish with no indication of BEQUALM liver pathology categories. • Prevalence of toxicopathic lesions generally >5% with prevalence of inflammatory lesions up to 100%. • High prevalence of FCA (>50%) of several types. Mean benign tumour prevalence of >15% (up to 25%) and malignant lesions more common but still relatively infrequent (up to 6%).

Figure 9.6. Site classification based on geographic location, levels of externally visible diseases and liver pathology categories.

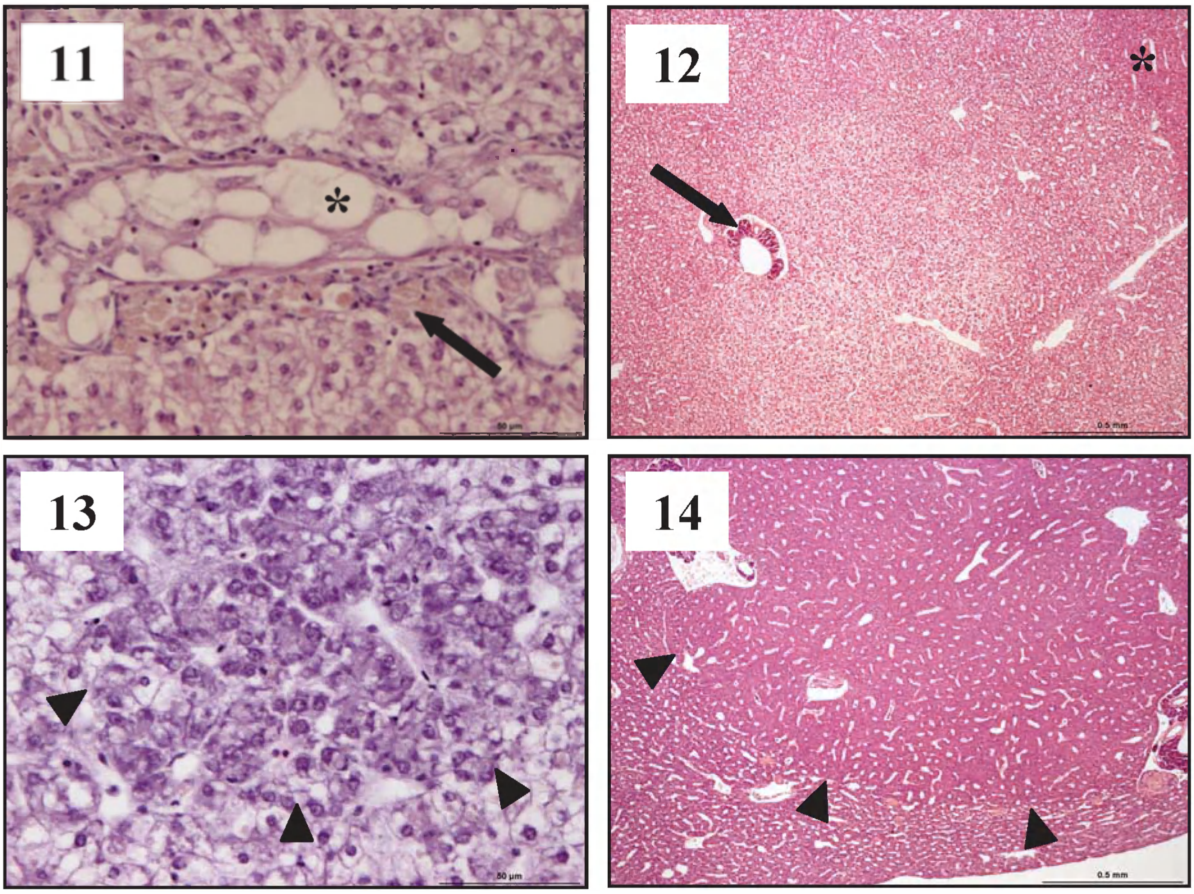


Figures 9.7 to 9.10. All are of dragonet liver tissue stained with haematoxylin and eosin. **Figure 9.7.** Section through a bile duct containing numerous *Myxidium incurvatum* parasites (arrow). Note the extensive vacuolation of the surrounding hepatocytes. Bar = 50 µm. **Figure 9.8**. Heavy infection with an unidentified coccidian parasite. Numerous bright pink eosinophilic sporocysts (C) can be seen throughout the section. Bar = 100 µm. **Figure 9.9**. Section showing a putative focus of cellular alteration (arrows). Normal hepatocytes (N). Bar = 100 µm. **Figure 9.10.** Hepatocellular adenoma (A) showing typical compression of the adjacent liver tissue (arrows). Bar = 100 µm.

9.3.4 Analysis of dragonet tissues

Sufficient numbers of dragonet were obtained from a number of sites in the Irish Sea to allow assessments of disease status and for sampling of tissues, bile and blood for pathology and bile contaminant metabolites and biomarker response respectively. External disease conditions were absent and most fish appeared to be in good condition. Occasional infections with the copepod parasite *Lernaeocera lusci*, sometimes in large numbers in the branchial cavity were noted. However, unlike in their gadoid hosts, parasitised fish were not emaciated.

Histological analysis of visceral tissues revealed a number of parasitic infections in the liver and kidney with myxozoan and coccidian parasites. Hepatic bile ducts were frequently occluded with plasmodia and spores of *Myxidium incurvatum* with little pathological response (Figure 9.7). Infections with an unidentified coccidian parasite within the hepatocytes resulted in significant destruction of the tissue in a few fish with most exhibiting low level infections with only few hepatocytes affected (Figure 9.8). In a few cases, apparent hepatocellular hydropic vacuolation may be attributable to coccidiosis, where parasite stages have migrated out of the liver leaving prominent vacuoles. In addition to pathology attributable to these parasites, there were putative toxicopathic lesions present in dragonets from several locations. Putative foci of cellular alteration (FCA) were seen in fish from Dundrum Bay, St Bees and West Lundy (Figure 9.9). Only few examples were detected and were not as discrete as FCAs occurring in flatfish. Further material is required to confirm their identity. A single case of hepatocellular adenoma (benign liver tumour) was recorded in a fish from SE Isle of Man (Figure 9.10). Focal necrosis and fibrosis of liver tissue was seen in two fish from St Bees. Renal pathology comprising of dilation of renal tubules and occasional degenerative glomerular changes were attributed to the presence of the myxozoan parasite *Davisia longibranchia* which was present in several of the fish sampled and in particular those from Dundrum Bay. No significant pathology was recorded in the gonads or spleen.



Figures 9.11 to 9.14. All are of plaice liver tissue stained with haematoxylin and eosin. **Figure 9.11.** Section through a bile duct showing vacuolation of the epithelium (*). Note the presence of pigment containing macrophage aggregates (arrow). Bar = 50 µm. **Figure 9.12.** Vacuolated focus of cellular alteration (vFCA), partially surrounding acinar pancreatic tissue (arrow). Part of a basophilic FCA can be seen (*). Bar = 0.5 mm. **Figure 9.13**. Section showing a small basophilic focus of cellular alteration (arrowheads). Bar = 50 µm. **Figure 9.14**. Hepatocellular adenoma (A) showing typical compression of the adjacent liver tissue (arrowheads). Bar = 0.5 mm.

9.3.5 Liver pathology in plaice

During 2005, a total of fifty plaice from Cardigan Bay were examined for the presence of hepatic pathology. A number of significant lesions types, including those employed for PAH specific monitoring purposes under the OSPAR Joint Assessment and Monitoring Programme (JAMP) were detected. Hydropic vacuolation of biliary epithelial cells was recorded in a single fish from inner Cardigan Bay (Figure 9.11). This pathology is classified as non-neoplastic toxicopathic in nature and has been recorded previously in plaice but is rarely seen in dab. Vacuolated and basophilic FCAs were also recorded in three fish. In one of these, both FCA types were present (Figure 9.12). In several cases these lesions were difficult to discern but in others small FCAs consisting of relatively few cells were readily detected, particularly the basophilic variants (Figure 9.13). A single case of hepatocellular adenoma was found in a plaice from northern Cardigan Bay (Figure 9.14). A variety of other conditions were present, including increased numbers of macrophage aggregates, focal inflammation and necrotic lesions as well as the presence of nematode infections and unidentified protistan parasites (material not shown).

9.3.6 Disease status of other species

Overall levels of disease in commercial fish were low (Table 9.3).

With the exception of whiting, numbers of other species examined were low and generally fewer than were examined in previous years (see Feist and Stentiford, 2005) and were insufficient to provide confidence in detecting low prevalence infections. Whiting were infected with a number of parasites and at some locations such as Amble almost all fish harboured nematodes. Other parasites such as *Cryptocotyle* and *Lernaeocera branchialis* were occasionally seen to be associated with emaciation of the host but in most cases fish appeared in good condition despite their parasite burdens. Very few cod were caught

Table 9.3. Disease status of non-target species from locations in the North Sea and Irish Sea sampled during 2004-2005.

Species	No. examined	Year	Location	Parasites/Pathology (No. affected)
Whiting	161	2004	Farne Deep	CR(5), LB(1)
"	126	"	Tees Bay	CR(5), NM(2)
"	134	"	Flamborough	CR(1), AC(3)
"	52	"	Off Humber	LB(11), AC(11)
"	100	2005	Amble	CR(1), LB(1), NM(99)
"	100	"	Dundrum Bay	CR(8), LB(2), DIC(22)
"	200	"	Thames Gabbard	CR(3), LB(14), NM(22), AC(25), LP(3), GL(2)
"	60	"	West Lundy	NM(2), EH(4)
Cod	9	2004	Farne Deep	NAD
"	2	"	Tees Bay	NM(1)
"	8	"	Amble	NAD
"	14	"	Off Humber	NM(5)
"	2	"	Smiths Knoll	NAD
Haddock	43	"	Tees Bay	NM(9)
"	6	"	Off Humber	NAD
"	66	"	Dundrum Bay	CR(3), LB(1)
"	100	2005	"	CR(9), SKD(1), ULC(2)
"	94	"	Flamborough	LB(15), NM(91), UC(5)
Plaice	50	"	Liverpool Bay	LP(29)
"	100	"	SE Isle of Man	NM(1), ULC(6), LP(11)
Pollack	93	"	Lyme Bay	CR(79), NM(49)
Bass	47	"	"	NM(6)

Key: NAD= No abnormalities detected
CR = *Cryptocotyle*
LB = *Lernaeocera branchialis*
NM = Nematodes (*Anisakis* spp.)
AC = *Acanthochondria* sp.
UC = Unidentified copepod
DIC = *Diclidophora merlangi*
LP = *Lepeophtheirus pectoralis*
GL = *Glugea* sp.
SKD = Skeletal deformity (scoliosis/lordosis)
ULC = Ulceration

and apart from nematode infections showed no evidence of disease. Sufficient haddock were collected from Dundrum Bay in the Irish Sea and Flamborough in the North Sea. Fish from Dundrum Bay harboured infections with *Cryptocotyle* at up to 9% prevalence in 2005, whereas haddock sampled during the same year from Flamborough were free of the infection. Conversely, 97% of haddock from Flamborough harboured nematode infections, whilst fish from Dundrum Bay (during both 2004 and 2005) were free from these parasites. During 2005 ulcerated plaice were recorded from southeast Isle of Man at 6% prevalence but none were seen in the fifty fish examined in Liverpool Bay. Infections with the external copepod parasite *Lepeophtheirus pectoralis* were seen in fish from both locations without evidence of adverse effects on the host. Pollack and bass examined from Lyme Bay on the south coast harboured nematode infections and 85% of bass were infected with varying numbers of *Cryptocotyle*.

9.4 Discussion

The health status of our native fish species provides an important indication of the overall health of our transitional waters and fully marine environments, particularly in consideration of disease conditions known to be associated with environmental contaminants. The role of the monitoring programme is to use changes in the prevalence of fish disease, to identify areas of potential concern where further in depth investigations may be needed, and in addition to detect the emergence of disease conditions. This approach is also supported by the use of biomarker and bioassay techniques in an integrated fashion to characterise the extent of the biological effects occurring. The ability to utilise long-term data sets to investigate disease trends and associations with a number of biological and environmental variables provides the means to test hypotheses on potential links between these factors (Wosniok *et al*., 2000). The data presented here indicate that the overall health status in dab is largely consistent with the levels recorded in previous years and for certain diseases, such as lymphocystis and epidermal hyperplasia/papilloma, the prevalence in most areas is low and probably not of concern. However, other conditions, including hyperpigmentation show increasing trends at several sites. The cause for this condition is currently unknown and recommendations from ICES WGPMO encourage investigations into its aetiology.

The levels of liver cancer in dab in Cardigan Bay appear to be increasing and are a cause for concern. More intensive investigations are currently underway to determine the extent of the liver pathology in dab and other flatfish populations in this area and to identify factors, which may be involved in the occurrence of the cancer. Lyons *et al*. (2006) reported such pathology in dab captured from sites in Cardigan Bay but biomarker and bioassay data did not indicate that adverse biological effects were occurring. Additionally, contaminant levels appear to be at relatively low levels in the areas sampled for dab.

In a previous study (Feist and Stentiford, 2005) we were able to apply multivariate statistics to fish disease data in order to achieve discrimination between sample locations in the North Sea and Irish Sea based on the presence and prevalence of external fish diseases and liver pathology, including cancerous lesions. A similar approach has been employed here for data collected during 2004 and 2005 fish disease monitoring programmes and has again demonstrated a consistent ability to discriminate sites from one another based upon the disease profile of fish from those sites. Using liver pathology data as a basis it has been possible to devise a site grading scheme based upon the severity of liver lesions observed at the site. Since we have shown that patterns of external disease conditions largely mirror those generated for liver pathology, these have also been taken into account in the typing scheme. Site typing moves us away from spatial comparisons and towards evidence-based comparisons of site-site similarity (eg fish from some Type C sites in the North Sea are very similar to fish from Type C sites in the Irish Sea). Site typing can then be used to find cross-correlates from biomarker and chemical data collected from the same fish and may allow us to identify specific risk factors that impact upon the ecosystem health status in UK waters.

Investigations on the occurrence of liver pathology in plaice from Cardigan Bay reported here, confirm that this species is susceptible to liver tumour formation. Previous studies have identified FCAs and adenoma in plaice from Carmarthen Bay and the Lyme Bay (Cefas, 2001). Simpson *et al*. (2002) did not detect any significant histopathological changes in the livers of plaice from the Mersey estuary and concluded that limited residence time in the estuary and small sample size may have reduced the chance of detecting toxicopathic pathology and tumour presence. It is therefore suggested that where sufficient numbers are caught, plaice provide an alternative species to dab and flounder for fish disease monitoring purposes, particularly since they also exhibit a range of external disease conditions. However, at present, differential susceptibility of individual species to liver cancer has not been investigated.

For integrated assessment purposes across the OSPAR maritime area, there are few fish species that are ubiquitous that could be used for biological effects monitoring purposes. OSPAR are developing guidelines for such integrated assessments using biological effects and chemical measurements via a number of workshops on integrated monitoring of contaminants and their effects in coastal and open sea areas (WKIMON). Within these, both plaice and dragonet are identified as alternative species to be used in monitoring. The results of the preliminary investigation presented here show that dragonet do not exhibit external disease conditions at sufficient prevalence for monitoring purposes but are susceptible to liver cancer and putative precursor lesions which may be contaminant related. In addition, sufficient numbers can be obtained from a variety of sites in British waters and it is possible to sample for multiple biomarkers from the same individual. As such, it seems likely that this species is suitable for monitoring purposes. Further studies are required to substantiate these preliminary findings and to ascertain the applicability of established biomarker methods eg. EROD, metallothionein, DNA adducts, for this species.

Data obtained for commercial fish species during 2004 and 2005 was rather limited, both in species range and numbers examined. The focus of the monitoring programme is on benthic flatfish species at established CSEMP locations. Despite the use of a Granton trawl, which captures both benthic and pelagic species, provides only limited numbers of commercial fish species for examination in the time available during the monitoring cruise. Nevertheless, compared to previous years a similar range of disease conditions and parasitic infections were observed. A newly recognised bacterial pathogen of farmed cod caused by *Franciscella* sp. (Olsen *et al*., 2006) caused as severe granulomatous pathology in several tissues including the liver and spleen. Although yet to be confirmed, this is highly likely to be the same disease, which has been seen in wild cod from the North Sea, since the mid-1980s, referred to as visceral granulomatosis (Cefas, 1998). There appears to be no risk to humans since the bacterium grows poorly at 30ºC and not at all at 37ºC on culture media.

Overall, diseases provide a definitive indication of health status in our commercial and non-commercial fish populations. The disease status of fish populations can be used as a reliable sentinel of marine health since the observable measure (ie the disease) is a direct result of a combined assault of the environment and potentially, of the pathogens within it. New approaches to considering disease at the level of the population are allowing us to compare the health status of apparently discrete populations of the same species from geographically distant locations. By correlating potential causal agents with the appearance of these diseases we move closer to identifying true impact of anthropogenic contaminants on the marine environment. Novel approaches to studying the genetic composition of fish populations also opens up possibilities of identifying differential susceptibility (eg to cancer) in our stocks - a feature that offers considerable interest to medical researchers studying similar cancers in human populations. The future of fish disease monitoring as a central tool in classifying health status in wild populations is clear but the true benefit of this approach will be better realised when data pertaining to contaminant biomarkers and to the contaminants themselves are fully integrated into analyses to assess the risk for their induction in wild animals (particularly the case for cancer). We are currently working towards such integration.

10. Eastern English Channel broadscale mapping

Author: Silvana Birchenough, David Limpenny and Ceri James*

10.1 Introduction

Every year approximately 22 million tonnes of marine sand and gravel are dredged from British waters (Singleton, 2001). The Eastern English Channel (EEC) contains substantial reserves, a proportion of which have been identified by the aggregates industry for future development and possible exploitation (Figure 10.1(a)). The area is known as the East Channel Region (ECR). The ECR is located in the centre of the eastern English Channel around 30 km off Beachy Head, and this area has not been previously licensed for marine aggregate extraction (Posford Haskoning, 2003). In addition the area possesses a variety of potentially vulnerable habitats, which might need protection to avoid damage resulting from the extraction of aggregate resources.

Six UK companies who are interested in exploiting resources form this area have formed a consortium known as the eastern English Channel Association (ECA). The ECA have sponsored an initial baseline survey and follow-up sampling programmes as part of a Regional Environmental Assessment (REA). This REA aims to characterise regional issues associated with resource and dredging management arising from the proposed developments. The REA outputs will help to characterize and augment the existing knowledge in the area prior to dredging.

Habitat mapping can be considered as a valuable tool to provide a spatial representation of discrete habitat units (Valentine *et al.*, 2004). Researchers have utilised biological and physical spatial information for generating habitat maps (Kostylev *et al.*, 2005; Brown *et al.*, 2001; Kostylev *et al.*, 2001). Early mapping efforts in the English Channel described and characterised the variability and distribution of benthic fauna inhabiting sediments over large geographical scales. Examples included the work conducted by Holme (1961, 1966), Sanvicente-Anorve *et al*. (1996) and Cabioch (1968) in French waters.

The British Geological Survey (BGS) have also undertaken surveys of the geology in the EEC. Results have shown that there is significant variability in geology in terms of bedrock and seabed sediments, which is likely to influence biotope variability. The area is composed of an extensive system of sediment infilled-channels incised into bedrock (Figure 10.1(c)) (Hamblin *et al.*, 1992). East of the infilled channels lie large-scale sandy bedforms and the presence of thin gravels can be found elsewhere (Figure 10.1(d)). Bedrocks are diverse ranging from chalk to tertiary muds and sand (Figure 10.1(b)), which exhibit different physical properties (ie, grain size, density, hardness and porosity, among others). Currently the relationship between the sediment, the underlying bedrock and habitats is unclear. Furthermore, there is also a lack of information on the relationship of habitats and sediment filled channels. It is important to understand the influence of the underlying geology on seabed habitats and also the influence of additional physical and biological processes are over a range of spatial scales.

The aims of this study are to create integrated habitat maps which will define the distribution of habitats, species and communities in the EEC, and to place them in a broad regional context. They will identify sites of biological, fisheries and geological significance and will highlight areas of high biodiversity that may require conservation under the EU Habitats Directive. This information will inform and support management of offshore resources. The preliminary results of seabed habitat mapping are presented in this section. Once completed, these results will provide essential information to contribute to the effective and sustainable management of this environment, for future commercial exploitation.

Additionally, this work will also help to create a wider assessment of the aggregate resources known to be present in the area and also to place into wider context the eastern English Channel region.

10.2 Methods

Four organisations are working on this multi-disciplinary programme. The overall project is lead by the BGS and is funded by Marine Aggregate Levy Sustainability Fund (MALSF) distributed under the Marine Environment Protection Fund (MEPF) administrated by Cefas.

BGS is responsible for the geophysical surveys and interpretations. Cefas is in charge of the collection of ground-truthing data sets (ie, biological, sediment samples, and additional physical data). Cefas is also leading on the biological analysis looking at the infaunal and epifaunal communities and their interaction with biotic and abiotic factors in collaboration with Marine Ecological Survey Ltd. (MESL). The Joint Nature Conservation Committee (JNCC) is working on the analysis of optical information (video sledge and still information) to characterise the species and habitats, which correspond to Annex I habitats known to be present in the area.

The area was sampled during 2005 and 2006 in order to complement the existing industry data sets with geophysical and ground-truthing surveys. BGS conducted a large-scale geophysical survey grid in 2005. This survey provided 'acoustic corridors' of multibeam bathymetry

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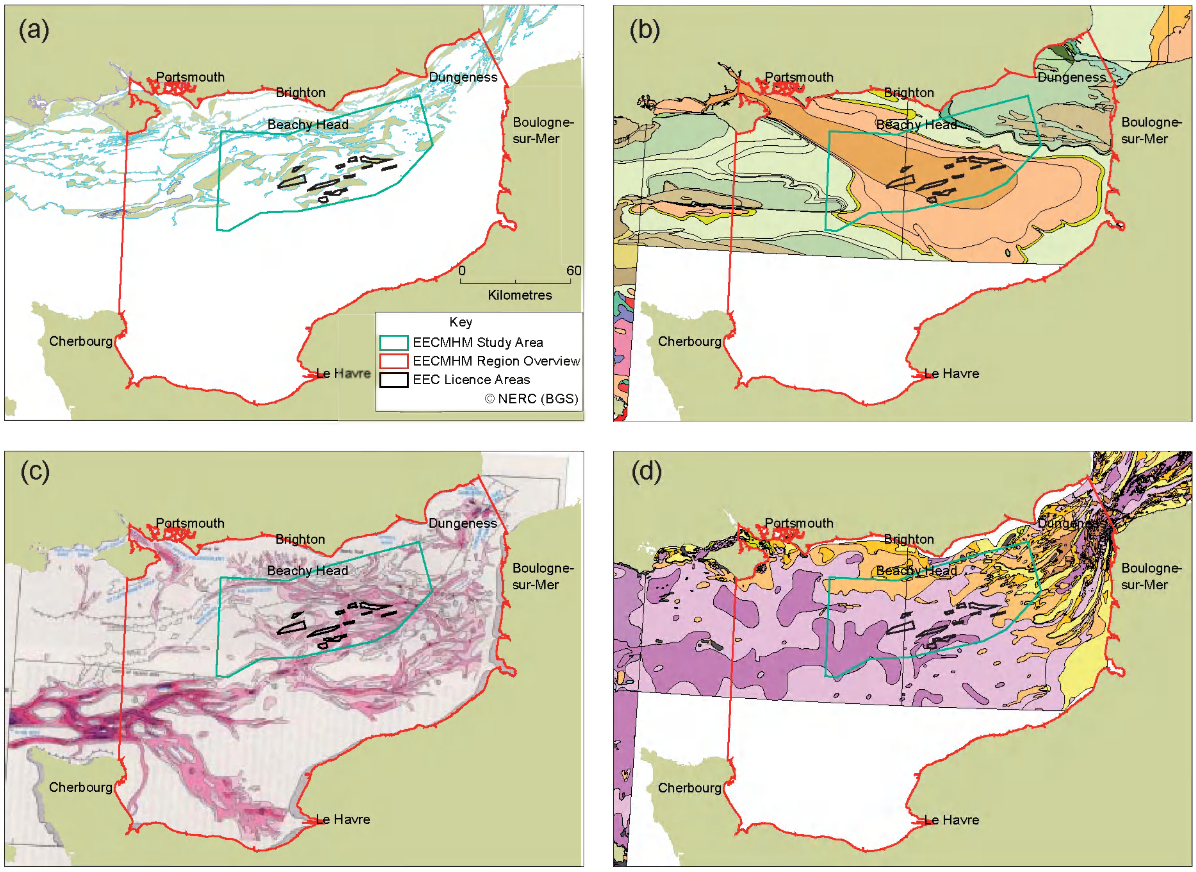


Figure 10.1. British Geological Survey (BGS) maps to describe: (a) proposed aggregate extraction licensed areas, (b) seabed geology characterisation showing chalk and tertiary muds, (c) seabed geology showing infilled sediment channels and (d) seabed sediments in the eastern English Channel region.

data and sidescan sonar covering approximately 7200 km^2. During 2006 a complementary geophysical survey was also conducted to collect additional acoustic information to complement existing lines to the east and west of the area was also undertaken.

Cefas conducted the ground-truthing survey over both years. Grabs, trawls and video-sledge transects were used to sample the areas where different habitat types were known to be present. This programme covered a total of 4850 km of geophysical lines, 230 grabs, 73 2-m beam trawl and 62 video and camera sites were collected over the two years (Figure 10.2).

The statistical analysis was conducted using univariate analyses and multivariate techniques were calculated from the Hamon grab samples to provide a quantitative assessment of the benthic fauna from within each acoustic region over time. The software PRIMER version 6 for Windows (Plymouth Routines in Multivariate Ecological Research) was used for this purpose. Non-parametric multi dimensional scaling (MDS) ordination using the Bray-Curtis similarity measure was applied to species abundance data following square root transformation of the data (excluding colonial taxa) to assess changes in species composition (Clarke and Warwick, 1994). Analysis of similarities (ANOSIM, Clarke, 1993) was performed to determine whether there were any significant differences in macrobenthic assemblage composition over time and between different acoustically distinct regions. The nature of groupings identified in the MDS ordinations was explored further by applying the similarity percentages programme (SIMPER) to determine the contribution of individual species towards the dissimilarity between years and stations.

The relationships between multivariate community structure and environmental variables will be assessed using the BIO-ENV routine. For simplicity only the initial MDS analysis is presented in this article for the distribution of infaunal communities.

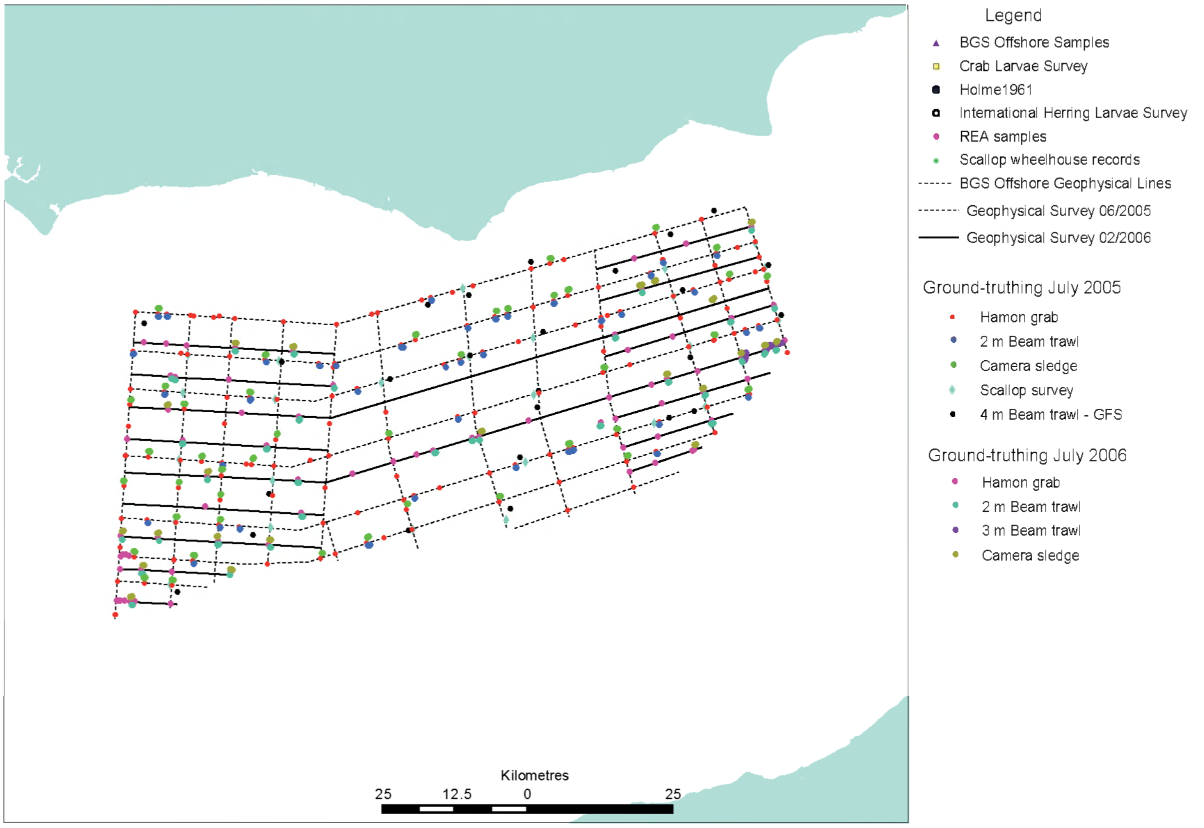


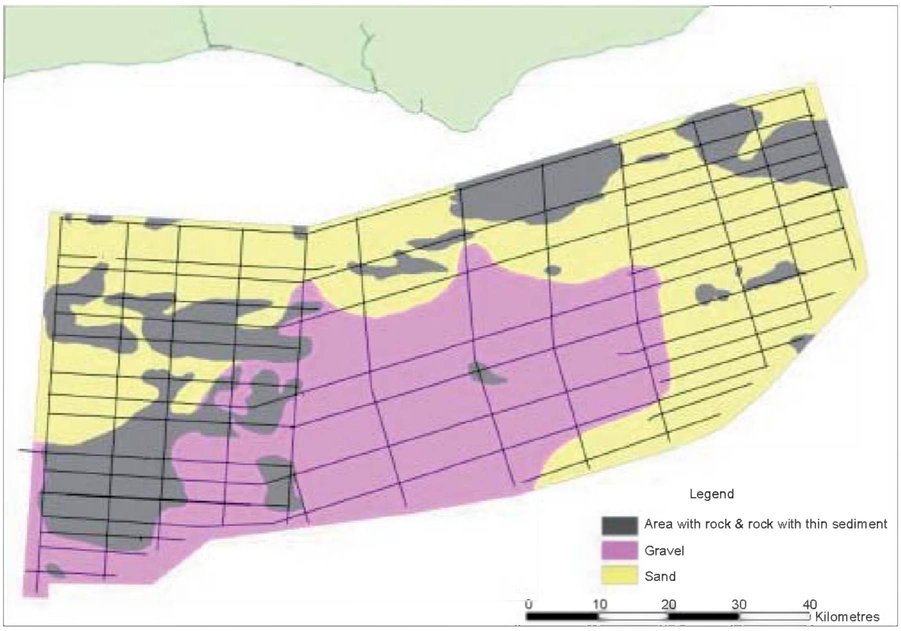
Figure 10.2. Sampling array for the completed geophysical and ground-truth survey for 2005 and 2006 in the Eastern English Channel.

10.3 Results

Geophysical interpretation

Preliminary analysis of the acoustic surveys and ground-truth samples indicated that the centre and west of the study area is composed of gravels. There is also a clear presence of sands distributed to the east, central and top of the west of the area. Rock and rock with thin sediments were encountered over the full study area but the majority can be seen in the west side of the study area (Figure 10.3).

Figure 10.3. Preliminary distribution of seabed sediments in the Eastern English Channel.



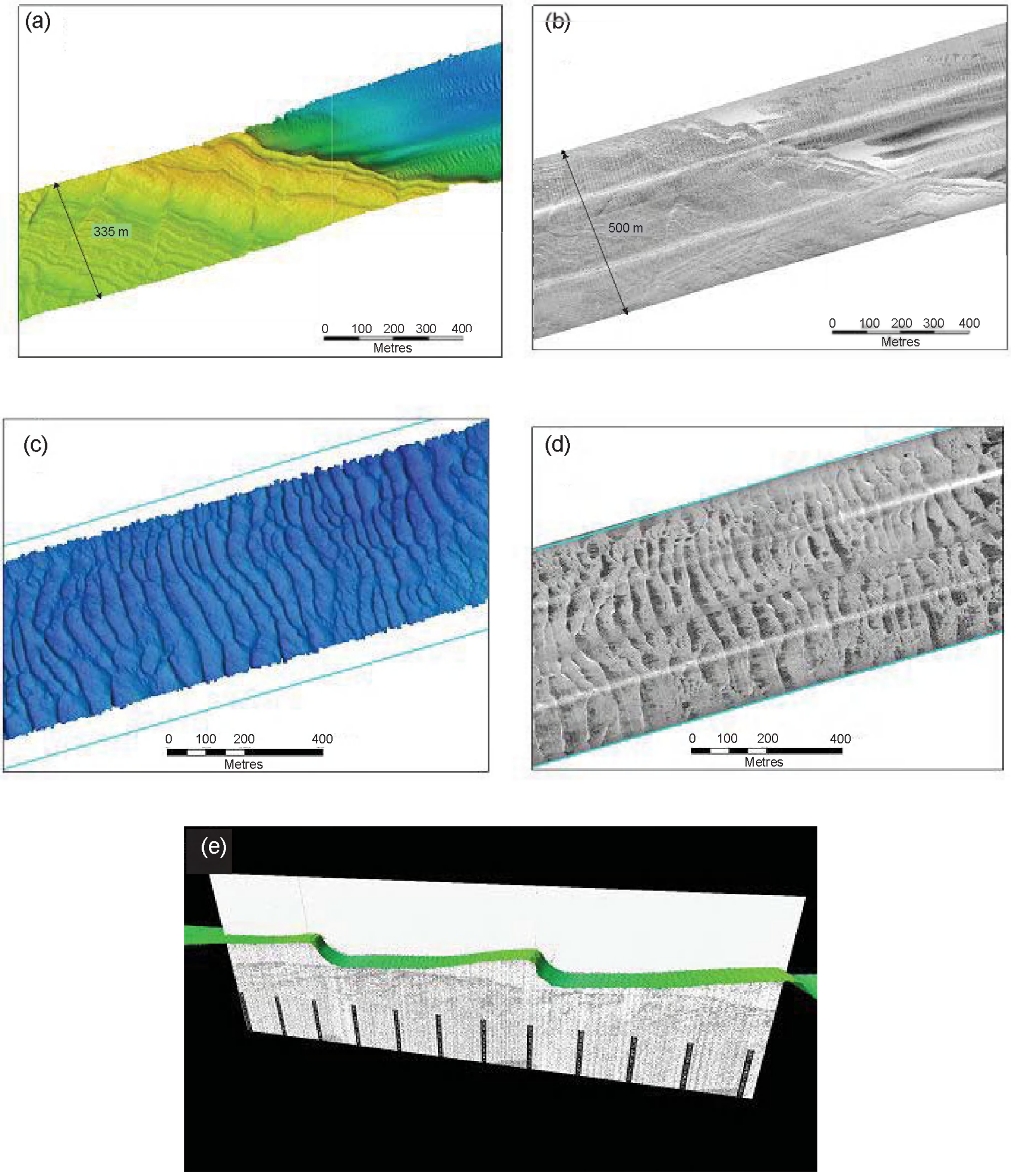


Figure 10.4. Examples of geophysical interpretation: (a) multibeam, (b) sidescan sonar of chalk and a 18 m high ridges and also the cross strike faulting, (c) multibeam image showing the presence of sand waves on gravelly sand, d) sidescan sonar showing the presence of sand waves on gravelly sand and (d) three-dimensional image of the character of the seabed across sand waves in rock.

The initial analysis of the geophysical interpretation showed wide range types of bedforms. A clear example can be seen in corridor 2, with a presence of chalk and a 18 m high ridges and also the cross strike faulting which might correspond to the presence of Annex I habitats (Figure 10.4(a-b)). Furthermore, the presence of sand wave was also observed in corridor 1 with a clear composition of sand and gravelly sand (Figure 10.4(c-d)). The multibeam information was also draped on boomer record and revealed the presence of sand waves on rock (Figure 10.4(e)).

10.4 Biological interpretation

Results the infaunal analysis accorded well with the initial BGS sediment map of the area. The MDS ordination demonstrated distinct clusters composed by groups J, I and L (Figure 10.5). These groups corresponded to communities that inhabit gravels, sandy gravels and sand respectively in the area. It is observed a clear transition of the species inhabiting these sediment types. Simper analysis helped to identify the species that typify these sediments. In the gravels the presence of *Galathea intermedia*, *Pomatoceros triqueter*, *Pisidia longicornis* and *Harmothoe* sp. were found among the top species. *Echinocyamus pusillus*, *Glycera lapidum*, Nemertea, *Aonides paucibranchiata* and *Spio filicornis* were abundant in the sandy gravels. In the sand the presence of *Echinocyamus pusillus*, *Poecilochaetus serpens*, Nemertea, *Spiophanes bombyx*, *Lagis koreni* and *Notomastus latericeus* were characteristics and abundant in their distribution.

The video survey also clearly demonstrated the presence of distinct habitats in the study area. Anemones and sponges are encrusted in cobbles (Figure 10.6(b)) in the western area. Brittle star beds were distributed in gravelly sediments in the western corner (Figure 10.6(a). Soft sediments hosting a less diverse epifauna community (Figure 10.6(d)) were found east of the area. Gravelly sediment was observed in the centre of the study area, where the industry have identified their resources with a much more diverse community and the presence of red gurnard *Aspitrigla cuculus* was identified from the video stills (Figure 10.6(c)).

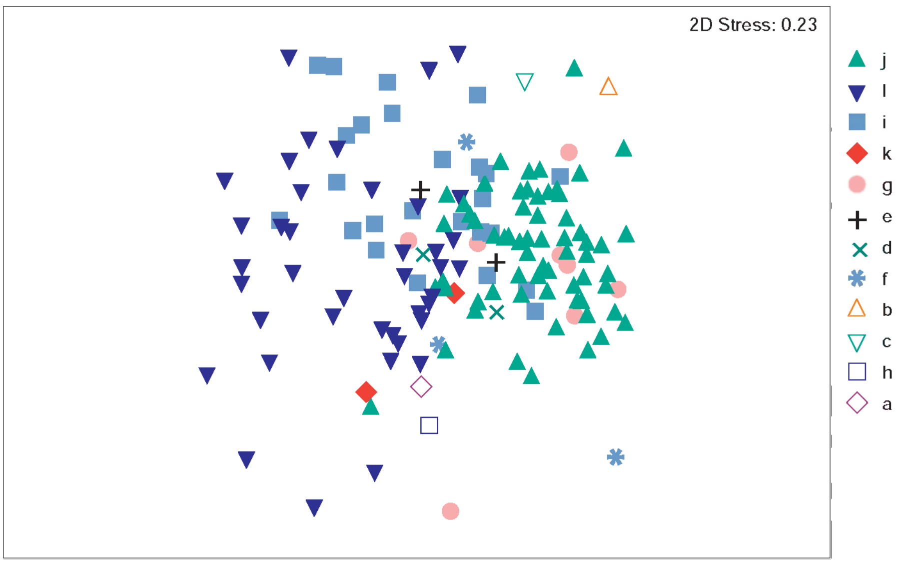
10.5 The way forward

This section provides preliminary result of this programme. The information from this ongoing multidisciplinary programme is due to finish in March 2007. The outcomes of this project will enhance our knowledge and understanding of the species, habitats and processes occurring at the ECR. The final results of this programme will be presented in the next AEMR.

The next steps to achieve the overall aims are:

- To complete the data analysis
- To generate GIS layers
- To produce classified maps
- To disseminate final project outputs

Figure 10.5. Multi-dimensional scaling (MDS) ordination showing the infaunal distribution at the eastern English Channel.



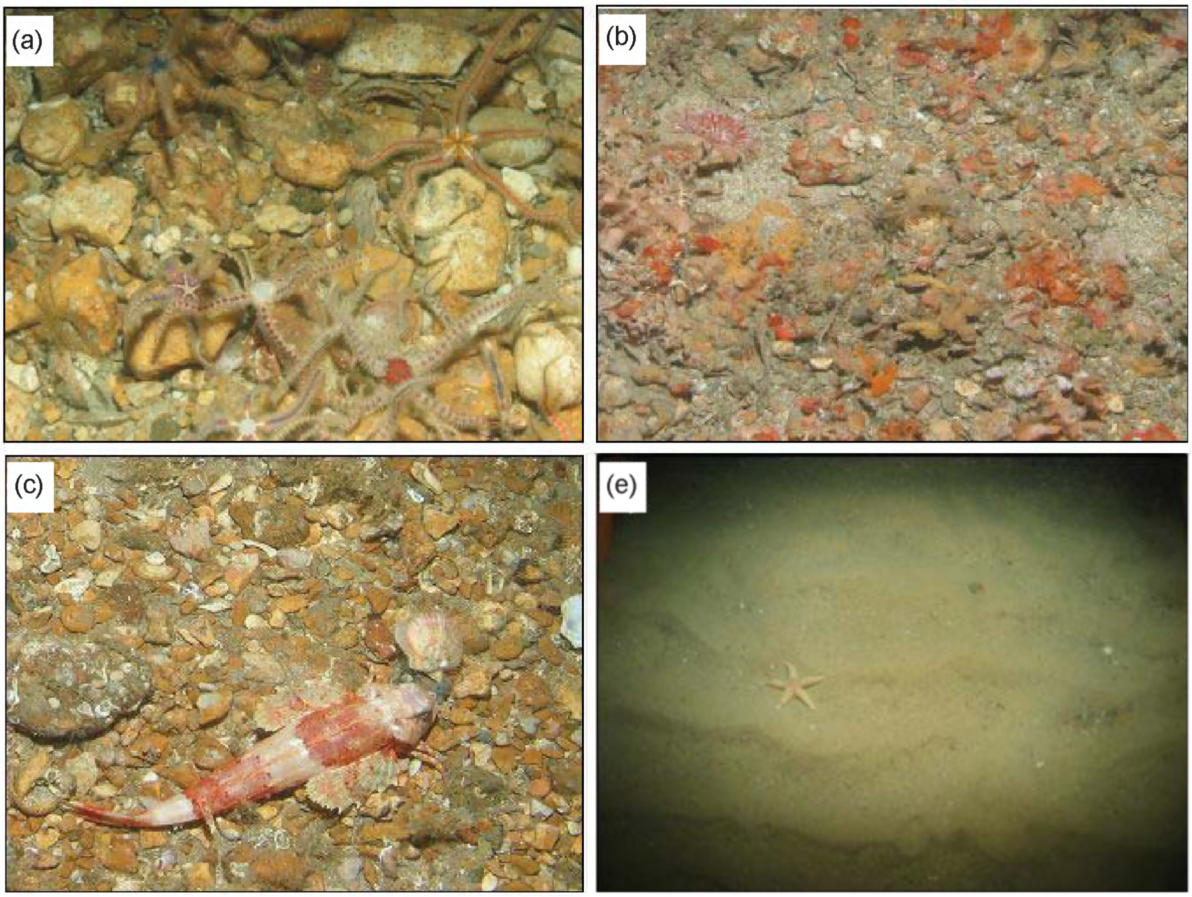


Figure 10.6. Video still images collected at the eastern English Channel: (a) brittlestars bed on coarse substratum, (b) sponges and anemones, (c) red gurnard *Aspitrigla cuculus* on gravel substratum and (d) starfish *Asterias rubens* on soft sediments.

The main outcomes of this programme will be:

- Critically review relevant scientific data for the EEC to identify knowledge gaps in the region. Collate the metadata within a GIS framework.
- Collect additional data through the conduct of new geophysical, sediment, biological and fisheries surveys in the EEC to target knowledge gaps.
- Integrate new and existing geological, geophysical and biological data within the GIS to provide comprehensive maps of the distribution of marine species and habitats within the EEC. This output will also assist in providing additional information on the distribution of any sensitive species or habitats.
- Identify causal relationships/correlations between the physical environment and associated fauna.
- To test and refine existing habitat classification systems.
- Provide additional geophysical, geological and biological data to underpin the integrated management of offshore resources in the EEC and to support improved spatial planning.
- Produce products that will be used to better manage marine offshore activities now and into the future and which will also resolve conflicts regarding seafloor use.
- Disseminate interpreted data, maps and new knowledge directly to stakeholders via the World Wide Web, reports, scientific publications, multimedia and other means.

Productive

11. Licensing of deposits in the sea

Author: Chris Vivian

11.1 Introduction

This section gives information about the licensing of deposits in the sea around the coasts of England and Wales in 2005 under Part II of the Food and Environment Protection Act 1985 (as amended) (FEPA) (Great Britain - Parliament, 1985a). In order to provide a complete picture for the UK as a whole, licensing statistics for Scotland and Northern Ireland are also included in this section.

11.2 Legislation and licensing authorities

The deposit of substances and articles in the sea, principally the disposal of dredged material (as opposed to discharge into the sea via pipelines) and the use of material during marine construction and coastal defence works, is controlled by a system of licences issued under Part II of FEPA. Certain operations (eg the deposit of scientific equipment or navigation aids) are exempt from licensing under the Deposit in the Sea (Exemptions) Order 1985 (Great Britain - Parliament, 1985b).

Following devolution in 1999, Defra (then MAFF) continued to license deposits in the sea around the Welsh coast on behalf of the Welsh Assembly Government. In Scotland, the licensing function became the responsibility of the Scottish Executive (then SERAD). In Northern Ireland the issuing of licences remained the responsibility of the Environment and Heritage Service, an agency of the Department of the Environment for Northern Ireland.

11.3 Enforcement

Scientists from the Cefas Burnham Laboratory have the powers to enforce Licence provisions. Visits are made to construction sites and disposal vessels. Samples are taken and records, including logbooks, are checked. Scientific staff carried out 21 inspections in 2005.

Officers of the Department's Marine Fisheries Agency (MFA) are charged with enforcing the provisions of FEPA (Part II) and undertake regular inspections from a network of port offices in England and Wales. The MFA carried out 153 inspections in 2005 in relation to construction works and the disposal of waste materials (dredged materials and a small amount of shellfish waste) at designated disposal areas. Further details are given in Table 11.1.

In England and Wales 3 written warning letters were issued for apparent breeches of licensing controls in 2005. Details are as follows:

Table 11.1. Inspection activity by the MFA during 2005.

District	No. of Inspections		No. of infringements
	Construction	Disposal	
Central	11	7	1
Eastern	34	2	6
London	0	0	0
Northern	6	8	1
South Eastern	25	7	2
South Western	13	5	1
Western	3	0	2
Wales	30	2	1
Annual Total	122	31	14

- Investigations into unlicensed construction works at Portland resulted in 2 official warnings being issued.
- Investigations into unlicensed construction works at Fleetwood resulted in an official warning being issued.

In England and Wales in 2005 there were 3 successful prosecutions for illegal marine works. The details are as follows:

- Investigations into unlicensed disposal and construction works at Mostyn, Flintshire resulted in a successful prosecution in 2005 where the defendant was fined £64,000 and ordered to pay costs of £24,000.
- Investigations into unlicensed disposal and construction works at Langstone Harbour, Hampshire resulted in 2 successful prosecutions in 2005 where the defendants were fined £8,000 and £5,000 respectively and were ordered to pay costs of £750 each.

In Scotland, certain authorised staff of the Fisheries Research Services (FRS) Marine Laboratory, Aberdeen and the Scottish Fisheries Protection Agency (SFPA) hold similar enforcement powers. The FRS made 12 enforcement visits in 2005. The SFPA made 3 enforcement visits in 2005.

In Northern Ireland the Environment and Heritage Service (EHS) made 10 enforcement visits in 2005. EHS also carried out 12 investigation visits in 2005 which resulted in 3 warning letters being issued to terminate the unlicensed activities.

Table 11.2. Summary of dredged material licensed and disposed of at sea in 2005.

Country	Year	Licences issued	Licensed quantity (tonnes)	Wet tonnage deposited	Dry tonnage deposited	Quantities of metal contaminants in wastes deposited (tonnes)						
						Cd	Cr	Cu	Hg	Ni	Pb	Zn
England and Wales	2001	124	39,297,549	29,660,448	14,881,254	7.61	1,040	731	5.84	478	1,099	3,310
	2002	124	72,851,190	27,884,495	14,725,603	5.53	912	457	4.66	409	1,166	2,664
	2003	97	31,836,123	29,526,580	15,800,897	5.41	950	498	4.29	443	1,183	2,694
	2004	80	44,790,919	28,516,645	14,949,123	5.27	886	513	4.35	412	1,190	2,648
	2005	87	37,483,750	27,777,496	14,792,977	5.18	764	579	4.30	419	1,037	2,617
Scotland	2001	29	3,307,800	2,217,981	1,162,856	0.36	79	48	0.74	36	77	165
	2002	21	2,959,045	2,203,016	1,188,129	0.33	59	46	0.85	29	69	134
	2003	29	3,573,981	2,764,020	1,647,881	0.61	70	57	1.40	41	101	175
	2004	23	2,412,670	1,484,408	742,204	0.19	27	19	0.51	14	31	54
	2005	20	5,293,220	2,723,703	1,376,334	1.23	181	112	1.59	69	174	360
Northern Ireland	2001	3	183,000	3,420,411	2,495,714	0.72	246	37	0.42	66	76	226
	2002	8	1,161,500	976,102	458,108	0.46	31	19	0.19	19	26	86
	2003	2	189,900	115,404	73,382	1.47	8	4	0.06	3	2	12
	2004	4	432,904	111,208	79,135	0.04	3	1	0.06	1	1	7
	2005	3	37,800	585,187	308,111	0.11	23	10	0.03	13	14	47
UK Total	2001	156	42,788,349	35,298,840	18,539,824	8.69	1,365	816	7.01	579	1,251	3,701
	2002	153	76,971,735	31,063,613	16,371,841	6.31	1,003	522	5.70	457	1,261	2,884
	2003	128	35,600,004	32,406,004	17,522,159	7.50	1,027	559	5.75	487	1,286	2,881
	2004	107	47,636,493	30,112,261	15,770,462	5.50	917	533	4.92	427	1,223	2,709
	2005	110	42,814,770	31,086,386	16,477,422	6.52	968	700	5.92	502	1,225	3,024

Notes: Tonnages deposited relate to quantities in the calendar year 2005, which may be covered by 2 or more licences, including one or more issued in previous years.

11.4 Licensing of dredged material

Table 11.2 give details for the period 2001 to 2005 of the number of sea disposal licences issued, the quantity of waste licensed and the quantity actually deposited, together with information on those contaminants in the wastes which the UK is required to report internationally to meet obligations under the OSPAR and London Conventions. A proportion of the trace metals in this dredged material is natural, but the mineral structure is such that it will not be available to marine organisms.

Figure 11.1 shows the main disposal sites used in 2005 and the quantities used at each site. Although applications for licences are required to show evidence that they have considered alternative disposal options including beneficial use, the problems of having silty materials, and matching the timing of dredging campaigns and the demand for sediments, have meant that most of the finer materials, in particular, are deposited at sea.

11.5 Other licensed activity

Under Part II of FEPA, licences are also required for certain other activities or deposits made below the mean high water springs mark for construction purposes. Each licence application is carefully considered, in particular, to assess the impact on the tidal and intertidal habitat, hydrological effects, potential interference to other users of the sea and risk to human health. Details of these licences issued in 2005 are shown in Table 11.3.

Further activities involve the use of tracers, the application of biocides, and burial at sea. Generally the anticipated environmental impact from these deposits is minimal and little or no monitoring is required. Details of these licences issued in 2005 are also shown in Table 11.3.

Such licences have also authorised the disposal of a small amount of fish waste, details given in Tables 11.4(a) and (b).

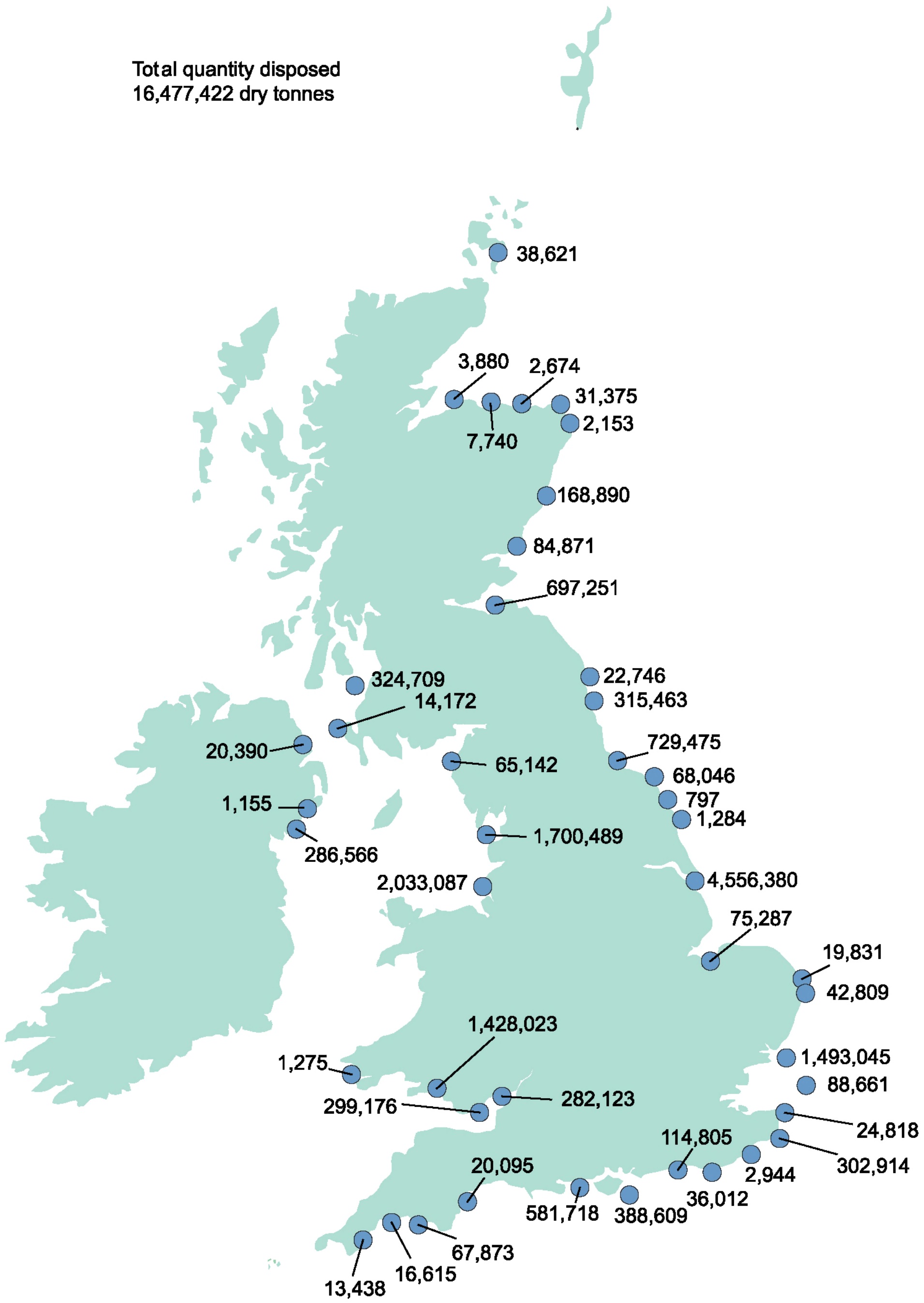


Figure 11.1. Amounts of dredged material disposed of at sea in 2005, in dry tonnes.

Table 11.3. Other categories of licences issued in 2005.

Licence category	England and Wales	Scotland	Northern Ireland	Total
Construction - new and renewal	274	148	13	435
Tracers, biocides etc.	8	2	0	10
Burial at sea	15	0	0	15

Table 11.4(a). Fish waste licensed for disposal at sea in 2005 [1].

Country	Licensed quantity (tonnes) [1]	Company and source of waste	Disposal sites	Quantity deposited (wet tonnes)	Quantity deposited (dry tonnes)
England and Wales	0	Quay Fresh and Frozen Foods Ltd, New Quay	New Quay	1,988	1,988

Notes : [1] No Fish Wastes were licensed or disposed of in Scotland or Northern Ireland during the period covered by this report.
For information on licensed quantities and tonnages deposited see footnote to Table 11.2.

Table 11.4(b). Summary of fish waste licensed and disposed of at sea in 2005.

Country	Year	Licences issued	Licensed quantity (tonnes)	Wet tonnage deposited	Dry tonnage deposited
England And Wales	2001	3	938	687	687
	2002	2	2,200	808	808
	2003	1	6,000	953	953
	2004	0	0	1,834	1,834
	2005	0	0	1,988	1,988
Scotland	2001	0	0	66	53
	2002	0	0	0	0
	2003	0	0	0	0
	2004	0	0	0	0
	2005	0	0	0	0
Northern Ireland	2001	0	0	0	0
	2002	0	0	0	0
	2003	0	0	0	0
	2004	0	0	0	0
	2005	0	0	0	0
UK Total	2001	3	938	753	740
	2002	2	2,200	808	808
	2003	1	6,000	953	953
	2004	0	0	1,834	1,834
	2005	0	0	1,988	1,988

Notes: For information on licensed quantities and tonnages deposited see footnote to Table 11.2.

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