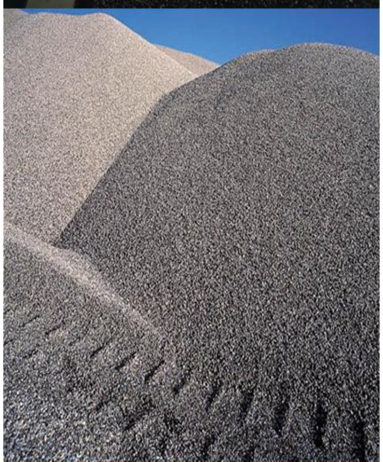

Marine Aggregate Dredging: A New Regional Approach to Environmental Monitoring

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Cover photos courtesy of BMAPA

ABSTRACT

The subject of this thesis is the marine aggregate dredging industry, and specifically the approach taken to the monitoring of environmental effects on the seabed. The thesis forms the evidence required to allow the author to be examined for a PhD by Publication, and comprises of a list of the author's publications, a confirmation of the author's contribution to the multi-authored papers, and a critical analysis of the published work. The critical analysis takes the form of an essay, in which a case for switching to a new system of environmental monitoring is outlined. The essay presents a logical development of ideas, starting with a description of the aggregates dredging industry. This is followed by a critical analysis of the author's past research, with a particular emphasis on how the findings from this work are relevant to the issue of monitoring. This earlier work has addressed themes of: Impact (Cumulative Effects), The Relationship between Sediments and Benthos, Recovery, Restoration, Habitat Mapping and Natural Variability. The essay then describes the current approach to monitoring, together with its limitations. This is followed by a description of the new monitoring approach, with an explanation of why it is considered more suitable for meeting the needs of both the industry and the industry regulators. The essay considers what steps would need to be taken to implement the approach in the major dredging regions of the UK.

PREFACE

Author's background

I am a benthic ecologist working at the Centre for Environment, Fisheries and Aquaculture Science (Cefas). In this role I undertake research on the effects of marine aggregate dredging on the seabed. I also provide technical advice to the aggregate industry regulators on matters relating to benthic ecology. This advice concerns marine licence applications to extract sand and gravel, and assessment of monitoring reports for active licensed areas. Over the last 11 years, I have authored a number of publications related to marine aggregate dredging, and it's my intention to submit this material for examination for a PhD by publication.

Purpose of the report

This report provides the evidence required by the University of East Anglia (UEA) to allow me to be examined for the degree. The report comprises three sections. Section A is a list of the publications on which the assessment is to be based. Section B is a confirmation of my contribution to the multi-author papers listed in section A. Section C is a critical analysis of the published works. The critical analysis takes the form of an essay in which I outline the case for the aggregate industry to switch to a new system of environmental monitoring. The case for switching is principally based on two recent publications specifically dealing with the new approach, but also on the earlier publications which underpin it. The essay is therefore an opportunity to critically appraise the entire body of published work.

Acknowledgements

I am grateful to my supervisor Professor Alastair Grant (UEA) for his comments on this thesis, and for the support he has provided during the course of my study. I would also like to thank the internal examiner, Dr Paul Dolman and the external examiner, Professor Mike Elliott (Institute of Estuarine and Coastal Studies) for their contribution to the examination process. I gratefully acknowledge Cefas for paying the examination fees, and for providing some time to allow me to write this report. I am also grateful to the Marine Management Organisation (MMO), Department for Environment, Food and Rural Affairs (Defra), The Crown Estate (TCE), and the British Marine Aggregate Producers Association (BMAPA) who have also provided funds, through a research project (Ref: C5922 - Development of Regional Seabed Monitoring Plans for the Marine Aggregates Industry), to assist in the completion of this report.

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[2] **Cooper, K.M.**, Burdon, D., Atkins, J.P., Weiss, L., Somerfield, P., Elliott, M., Turner, K., Ware, S., Vivian, C., 2013. Can the benefits of physical seabed restoration justify the costs? An assessment of a disused aggregate extraction site off the Thames Estuary, UK. Marine Pollution Bulletin 75, 33-45.

[3] **Cooper, K.M.**, 2012. Setting limits for acceptable change in sediment particle size composition following marine aggregate dredging. Marine Pollution Bulletin 64, 1667-1677.

[4] Wan Hussin, W.M.R., **Cooper, K.M.**, Barrio Froján, C.R.S., Defew, E.C., Paterson, D.M., 2012. Impacts of physical disturbance on the recovery of a macrofaunal community: A comparative analysis using traditional and novel approaches. Ecological Indicators 12, 37-45.

[5] **Cooper, K.M.**, Curtis, M., Wan Hussin, W.M.R., Barrio Froján, C.R.S., Defew, E.C., Nye, V., Paterson, D.M., 2011. Implications of dredging induced changes in sediment particle size composition for the structure and function of marine benthic macrofaunal communities. Marine Pollution Bulletin 62, 2087-2094.

[6] Barrio Froján C.R.S., **Cooper, K.M.**, Bremner, J., Defew, E.C., Wan Hussin, W.M.R., Paterson, D.M., 2011. Assessing the recovery of functional diversity after sustained sediment screening at an aggregate dredging site in the North Sea. Estuarine, Coastal and Shelf Science 92, 358-366.

[7] **Cooper, K.M.**, Ware, S., Vanstaen, K. and Barry, J., 2011. Gravel seeding - A suitable technique for restoration of the seabed following marine aggregate dredging? Estuarine, Coastal and Shelf Science 91, 121-132.

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[9] Barrio Froján C.R.S., Boyd, S., **Cooper, K.M.**, Eggleton, J.D., Ware, S., 2008. Long-term benthic responses to sustained disturbance by aggregate extraction in an area off the east

coast of the United Kingdom. Estuarine, Coastal and Shelf Science 79, 204-212.

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[11] **Cooper, K.M.**, Boyd, S.E., Aldridge, J., Rees, H.L., 2007. Cumulative impacts of aggregate extraction on seabed macro-invertebrate communities in an area off the east coast of the United Kingdom. Journal of Sea Research 57, 288-302.

[12] Boyd, S.E., Limpenny, D.S., Rees, H.L., **Cooper, K.M.**, 2005. The effects of marine sand and gravel extraction on the macrobenthos at a commercial dredging site (results 6 years post-dredging). ICES Journal of Marine Science 62, 145-162.

[13] Brown, C.J., Hewer, A.J., Meadows, W.J., Limpenny, D.S., **Cooper, K.M.**, Rees, H.L., 2004a. Mapping seabed biotopes at Hastings Shingle Bank, Eastern English Channel. Part 1: Assessment using Sidescan sonar. Journal of the Marine Biological Association 84, 481-488.

[14] Brown, C.J., Hewer, A.J., Meadows, W.J., Limpenny, D.S., **Cooper, K.M.**, Rees, H.L., 2004b. Mapping seabed biotopes using sidescan sonar in regions of heterogeneous substrata: Case study east of the Isle of Wight, English Channel. Underwater Technology 26 (1), 27-36.

[15] Boyd, S.E., Limpenny, D.S., Rees, H.L., **Cooper, K.M.**, Campbell, S., 2003. Preliminary observations of the effects of dredging intensity on the recolonisation of dredged sediments off the southeast coast of England Area 222. Estuarine, Coastal and Shelf Science 57, 209-223.

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This study was conceived by Keith, and followed on from the restoration experiment described in Cooper et al. (2011)[7]. Keith managed the project and undertook the assessment of physical impacts, identification of possible restorative techniques, and costs of restoration. Socio-economic aspects of the work were undertaken by co-authors from the Institute of Estuarine and Coastal Studies (IECS) and the University of East Anglia (UEA).

I confirm that the above paragraph is a true reflection of the contribution made by Keith Cooper to this paper.

Daryl Burdon (15/09/2013)

[4] Wan Hussin, W.M.R., **Cooper, K.M.**, Barrio Froján, C.R.S., Defew, E.C., Paterson, D.M., 2012. Impacts of physical disturbance on the recovery of a macrofaunal community: A comparative analysis using traditional and novel approaches. Ecological Indicators 12, 37-45.

The lead author on this paper (Dr Wan Hussin) was a PhD student co-supervised by Keith Cooper and David Paterson (University of St Andrews). The paper dealt with the issue of seabed recovery, by extending a time series dataset from extraction Area 222 (see Boyd et al., 2003[15], 2005[12]). The paper also assessed recovery in terms of ecosystem functioning, a similar exercise to that carried out in Cooper et al. (2008)[8]. The survey work was planned and led by Keith. Sample processing and data analyses were undertaken by Wan Hussin, under the direction of Keith. The preparation of the paper was led by Wan Hussin, with Keith providing comments on various drafts.

I confirm that the above paragraph is a true reflection of the contribution made by Keith Cooper to this paper.

David Paterson (29/05/2013)

[5] **Cooper, K.M.**, Curtis, M., Wan Hussin, W.M.R., Barrio Froján, C.R.S., Defew, E.C., Nye, V., Paterson, D.M., 2011. Implications of dredging induced changes in sediment particle size composition for the structure and function of marine benthic macrofaunal communities. Marine. Pollution. Bulletin 62, 2087-2094.

This study was designed by Keith, and extended the work undertaken in Barrio Froján et al. (2011)[6] and Cooper et al. (2008)[8]. Co-authors were given responsibility for analysing data using different methods. The Anosim values from the comparison of treatment and reference conditions for each site were supplied to Keith who undertook the final analysis using a weighted sensitivity Index. All figures, tables and text were prepared by Keith. Co-authors provided comments on draft versions of the paper.

I confirm that the above paragraph is a true reflection of the contribution made by Keith Cooper to this paper.

David Paterson (29/05/2013)

[6] Barrio Froján C.R.S., **Cooper, K.M.**, Bremner, J., Defew, E.C., Wan Hussin, W.M.R., Paterson, D.M., 2011. Assessing the recovery of functional diversity after sustained sediment screening at an aggregate dredging site in the North Sea. Estuarine, Coastal and Shelf Science 92, 358-366.

This study, conducted under a research project developed and led by Keith, utilised a recovery dataset from the Area 408 extraction site. Macrofaunal data from Area 408 were acquired by Keith under a previous study. The paper broadly follows the methodology adopted in Cooper et al. (2008)[8]. Keith had responsibility for the analysis of data using two of the five techniques used (Biological Traits analysis and Production). He also commented on earlier drafts of the manuscript.

I confirm that the above paragraph is a true reflection of the contribution made by Keith Cooper to this paper.

David Paterson (29/05/2013)

[7] **Cooper, K.M.**, Ware, S., Vanstaen, K. and Barry, J., 2011. Gravel seeding - A suitable technique for restoration of the seabed following marine aggregate dredging? Estuarine, Coastal and Shelf Science 91, 121-132.

The idea to investigate gravel seeding as a potential means of restoring a coarse sediment habitat was Keith's. He had overall responsibility for selecting the study site, and for designing the seeding strategy and follow up surveys. Keith was present on the dredger whilst the seeding operation took place, and he led the subsequent monitoring of the site. Sample processing was undertaken by Keith and Suzanne Ware. The analysis of data, production of figures and tables, and the writing of the paper were all completed by Keith, with input from Suzanne Ware.

I confirm that the above paragraph is a true reflection of the contribution made by Keith Cooper to this paper.

Dr Suzanne Ware (29/05/2013)

[8] **Cooper, K.M.**, Barrio Froján, C.R.S., Defew, E., Curtis, M., Fleddum, A., Brooks, L. and. Paterson, D.M., 2008. Assessment of ecosystem function following marine aggregate dredging. Journal of Experimental Marine Biology & Ecology 366, 82-91.

The idea for this study was initiated by Keith, following a discussion with David Paterson. Specifically, Keith wanted to analyse the faunal recovery data from Cooper et al. (2007)[10] from the perspective of ecosystem function. Keith tasked St Andrews University with leading the identification of possible functional analysis techniques. Keith undertook data analysis for two of the five techniques, with co authors responsible for the remaining three techniques. All tables, figures, and an initial draft of the paper were produced by Keith. Latterly, Christopher Barrio Froján finalised the manuscript.

I confirm that the above paragraph is a true reflection of the contribution made by Keith Cooper to this paper.

Prof David Paterson (29/05/2013)

[9] Barrio Froján C.R.S., Boyd, S., **Cooper, K.M.**, Eggleton, J.D., Ware, S., 2008. Long-term benthic responses to sustained disturbance by aggregate extraction in an area off the east coast of the United Kingdom. Estuarine, Coastal and Shelf Science 79, 204-212.

This paper was written under a project led by Keith. This paper investigates the temporal variability of the benthic communities initially identified in Cooper et al. (2007)[11]. Keith had responsibility for the collection and processing of the majority of samples. He also produced the map, and provided comments on the earlier drafts of the paper.

I confirm that the above paragraph is a true reflection of the contribution made by Keith Cooper to this paper.

Dr Christopher Barrio Froján (29/05/2013)

[10] **Cooper, K.M.**, Boyd, S.E., Eggleton, J.E., Limpenny, D. S., Rees, H. L., and Vanstaen, K., 2007. Recovery of the seabed following marine aggregate dredging on the Hastings Shingle Bank off the southeast coast of England. Estuarine, Coastal and Shelf Science 75, 547-558.

The experimental design for this work was a collaborative effort involving Boyd, Limpenny and Cooper. Annual monitoring surveys were designed and led by Keith. He also oversaw the processing of macrofaunal and sediment samples. Keith was responsible for the analysis of data, production of tables and figures, and the writing of the paper.

I confirm that the above paragraph is a true reflection of the contribution made by Keith Cooper to this paper.

Dr Sian Limpenny (née Boyd) (29/05/2013)

[11] **Cooper, K.M.**, Boyd, S.E., Aldridge, J., Rees, H.L., 2007. Cumulative impacts of aggregate extraction on seabed macro-invertebrate communities in an area off the east coast of the United Kingdom. Journal of Sea Research 57, 288-302.

This study used an existing dataset for macrofauna and sediments to investigate whether there was evidence of a broadscale cumulative impact associated with aggregate dredging. It was Keith's idea to assign samples to treatment categories (Direct impact zone, Indirect impact zone, Reference zone) using the dredger's 'black box' Electronic Monitoring System (EMS) data for the direct impact zone, and a modelled area for the indirect impact zone. Assigning samples to these treatment categories allowed the study to demonstrate a potential cumulative effect of dredging. The use of EMS and modelled secondary impact zones is now widespread in survey design, and this study showed how it could be done. All data analyses, with the exception of the generation of the modelled indirect zone, were undertaken by Keith. The paper's text, figures and tables were also produced by Keith, with the exception of section 2.3.2 which deals with the modelling approach. Boyd and Rees provided comments on earlier drafts of the manuscript.

I confirm that the above paragraph is a true reflection of the contribution made by Keith Cooper to this paper.

Dr Sian Limpenny (née Boyd) (29/05/2013)

[12] Boyd, S.E., Limpenny, D.S., Rees, H.L., **Cooper, K.M.**, 2005. The effects of marine sand and gravel extraction on the macrobenthos at a commercial dredging site (results 6 years post-dredging). ICES Journal of Marine Science, 62: 145-162.

The experimental design for this work was a collaborative effort involving Boyd, Limpenny and Cooper. Keith was responsibile for the design and conduct of survey work. He was also responsible for overseeing the processing of samples, and for the integration of data from different years. Data analysis and initial drafting of the paper were undertaken by Boyd, with comments provided by Keith and other co-authors at various stages of the process.

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Dr Sian Limpenny (née Boyd) (29/05/2013)

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Keith contributed to the design and conduct of survey work. He was responsible for the processing of macrofaunal samples, and he provided comments on earlier drafts of the paper.

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Mr David Limpenny (29/05/2013)

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Mr David Limpenny (29/05/2013)

MARINE AGGREGATE DREDGING: A NEW REGIONAL APPROACH TO ENVIRONMENTAL MONITORING

1. INTRODUCTION

1.1 PURPOSE

This critical analysis takes the form of an essay. The subject of the essay is the marine aggregate dredging industry, and specifically the approach taken to monitoring the environmental impacts of dredging during the lifetime of operations. I will address this subject in two ways. Firstly, through an examination of the literature, focussing on those papers I have personally authored. These papers should be read in conjunction with this essay. Secondly, I will draw upon my experience as a technical advisor to the industry regulators. My aim is to demonstrate that the current approach to monitoring the environmental impacts of dredging on the seabed should be replaced by a new approach. Whilst the new approach is the subject of two of my recent papers (Cooper, 2012[3]; Cooper, 2013[1]), its rationale relies on the findings from my earlier research (Brown et al., 2002[16], 2004a[13], 2004b[14]; Boyd et al., 2003[15], 2005[12]; Cooper et al., 2007a[11], 2007b[10], 2008[8], 2011b[5], 2011a[7], 2013[2]; Barrio Froján et al., 2008[9], 2011[6]; Wan Hussin et al., 2012[4]). Due to their supporting role, I have chosen to present the papers in a reverse chronological order. This essay is intended to form a critical analysis of this entire body or work.

1.2 ESSAY STRUCTURE

I begin with a brief overview of the marine aggregate dredging industry, to provide a context for readers unfamiliar with the subject. This is followed by a description of the research that I have undertaken over the last 11 years, with a particular emphasis on its relevance to the monitoring of environmental impacts, and the management of the industry. I then provide an overview of the current approach to monitoring, and highlight why I believe it is no longer fit-for-purpose. This section is followed by a description of the proposed new monitoring approach, together with a review of the underpinning science. I discuss the advantages and disadvantages of the new system before considering how it could be implemented. I end the essay with a conclusion.

2. THE AGGREGATES INDUSTRY

2.1 WHAT IT DOES

The marine aggregates industry supplies sand and gravel from the seabed for use in construction, fill and coastal defence (Highley et al., 2007). In the UK, the industry supplies around 20 Mt of material each year. This constitutes around 20% of the demand, with the remainder coming from land based sources (BMAPA, no date, (a)). It is anticipated that marine sources of sand and gravel will play an increasingly important role as pressures on land based sources of material rise; a result of resource depletion, planning constraints, and environmental pressures (Highley et al., 2007).

2.2 EXTRACTION METHODS

Material is extracted from the seabed using purpose built suction hopper dredging vessels. The main features of a typical suction hopper dredger are shown in Figure 1. When in position, the vessel's dredge pipe is lowered by cables into the water. The seawater pump is then activated and water is drawn up the pipe. From the pipe, water passes up and over the screening tower and into the vessel hold. Once the drag head makes contact with the seabed, a mixture of water and sand/gravel is drawn up the dredge pipe and up to the screening tower. Heave compensation is used to ensure that the draghead maintains contact with the seabed. Once on the screening tower, the mixture of water and sediment can be passed over a grid which serves to separate one size of material from another. Rotation of the screening towers allows one size of material to pass into vessel hold, whilst the other is rejected over the side of the vessel. This process is known as 'screening'. Gradually the hold will fill with the water and sediment mix. Reject chutes (also known as spillways), located at the top of the vessel hold, allow for water and fine sediments to 'overspill' back into the sea. Once the hold is full with sediment, the ship will return to a dedicated wharf facility to offload its cargo. These facilities are generally located close to the point of use of the material, thereby reducing the costs of onward transport.

There are two methods of dredging used to extract sand and gravel in the UK: trailer suction hopper dredging, and anchor suction hopper dredging. The most common method employed in the UK is trailer dredging, although anchor dredging is sometimes used to target deep, localised deposits. The UK has a fleet of 28 dredgers which are operated by various companies (Highley et al., 2007). The size of these vessels varies, but a typical dredger will hold 5000 tonnes of material. The effective depth of operation is dependent on the size of the vessel, with the largest vessels capable of operating to 60m.

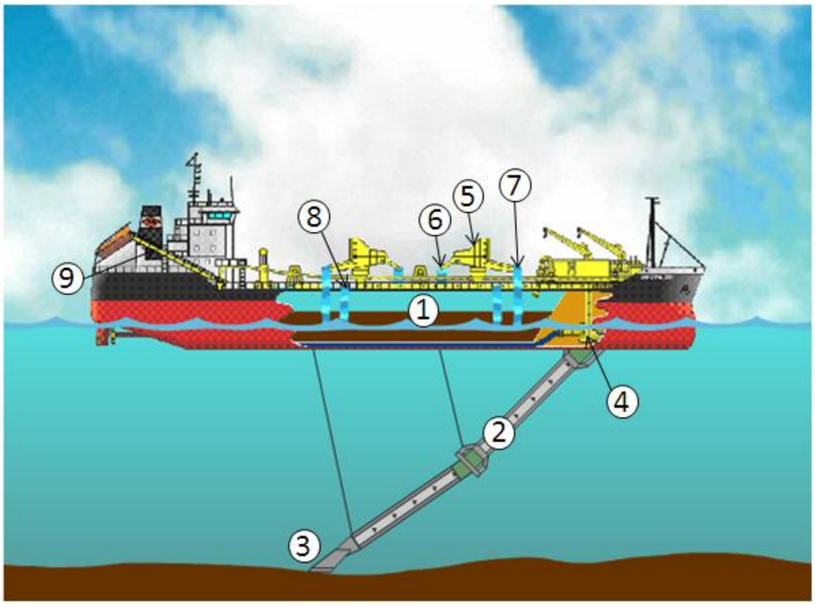


Fig. 1. Main features of a trailer suction hopper dredger including: (1) vessel hold, (2) dredge pipe, (3) drag head, (4) seawater pump, (5) screening tower, (6) retained material from screening, (7) rejected material from screening, (8) spillway, and (9) offloading conveyer. Background image courtesy of the British Marine Aggregate Producers Association.

2.3 AREA OF OPERATION

Material is currently extracted from around 70 licensed extraction areas located around the coast of England and Wales (Russell, 2011). These areas are located in seven dredging regions including the Humber, East Coast, Thames, Eastern English Channel, South Coast, Bristol Channel and North-west (Figure 2). Gravel and sand cargoes are targeted from the south and east coasts, while mainly sand is taken from the Bristol Channel and North-west regions. These differences result from local market requirements, but also from differences in the nature of the deposits between the regions (see Section 2.4). In 2010, the area of seabed licensed for extraction was 1291 km^2, although typically only a relatively small percentage of this area is dredged in any one year (BMAPA, no date (b)).

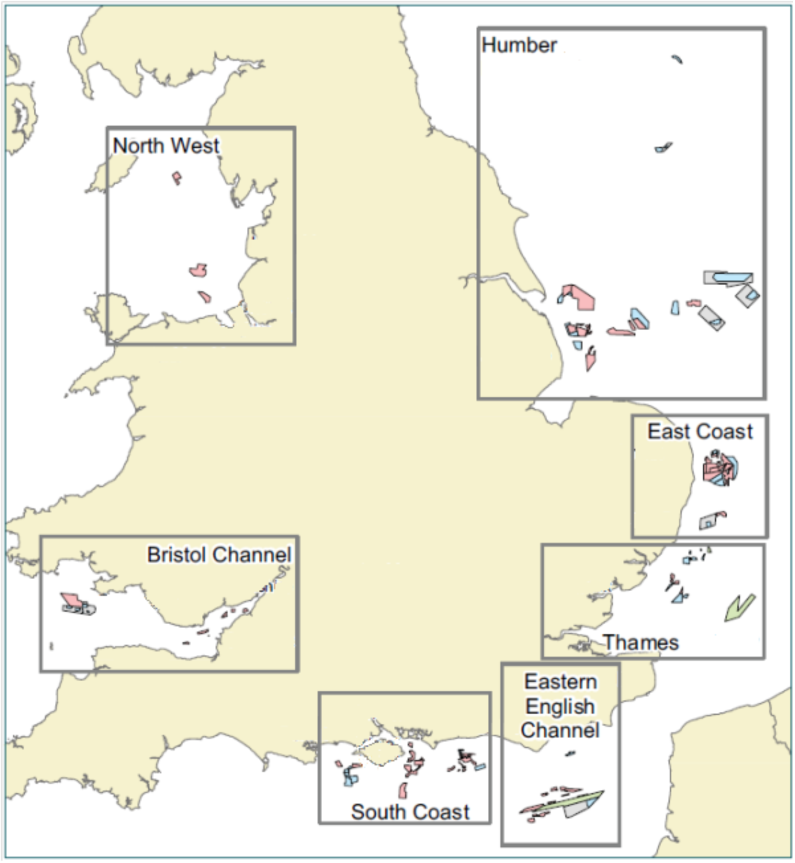


Fig. 2. Location of the main aggregate dredging regions (large rectangles). Individual extraction areas are coloured according to their status at the time the map was produced: Pink – active licence area, Grey – prospecting area, Blue – application area, Green – option area. Figure adapted from BMAPA (2011).

2.4 ORIGIN OF MATERIAL

The sand and gravel deposits targeted by the marine aggregates industry can be classified as either 'relict' or 'modern' (Velegrakis et al., 2010). Relict deposits were formed in the past under different environmental conditions. Modern deposits are formed and controlled by modern hydro and sediment dynamic conditions. Most exploitable deposits originate from the Quaternary, the geological period extending from 2.6 million years ago to present day. The Quaternary is subdivided into Pleistocene and Holocene epochs. The Pleistocene was characterised by repeated glaciations, and spanned the period from 2.6 MY to 11,700 years before present. In contrast, the Holocene is a warm interglacial period which began 11,700 years ago and continues today. During the Pleistocene sea levels were much lower, exposing much of the current continental shelf. Periglacial rivers drained across this landscape, and sand and gravel deposits are found in association with these historic river features. When sea levels eventually rose, these fluvial deposits were transformed into estuarine and eventually marine environments (Figure 3). As a result, the drowned or palaeovalleys can be characterised by fine grained sediments at the surface. Deposits are also associated with beaches which have been transgressed by rising sea level. Such material is typically well sorted and resistant to abrasion, making it particularly well suited for beach replenishment (Velegrakis, 2010). Lastly, there are pro-glacial till deposits comprising sandy gravels and coarse gravely sands. These deposits are formed by melt-water rivers at the head of a glacier or ice sheet (e.g. The Cleaver Bank in the Dutch sector (see van Moorsel, 1994), and the UK East Coast and Humber region deposits).

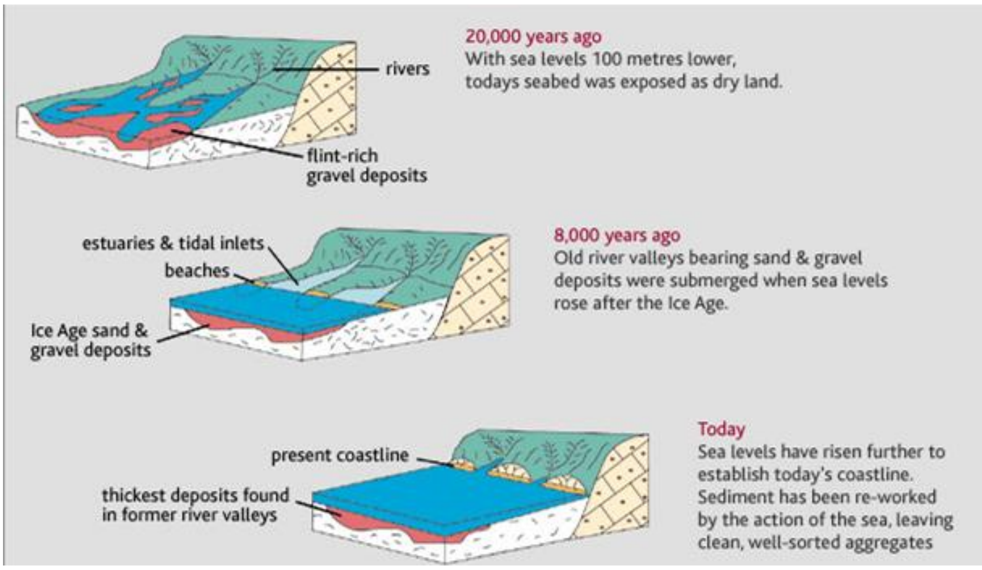


Fig. 3. Origin of relict marine aggregate deposits. Figure taken from BMAPA (no date (c)).

According to Velegrakis et al. (2010), modern deposits of interest to the marine aggregates industry generally comprise of mobile stores of sediment, including sand sheets, sand patches, sand banks, sand ribbons and subaqueous dunes. Of these, the sand banks and subaqueous dunes (see Figure 4) are most readily exploited. The targeting of these particular deposits stems from their relative thickness; sand sheets and sand ribbons are by their nature thinner.

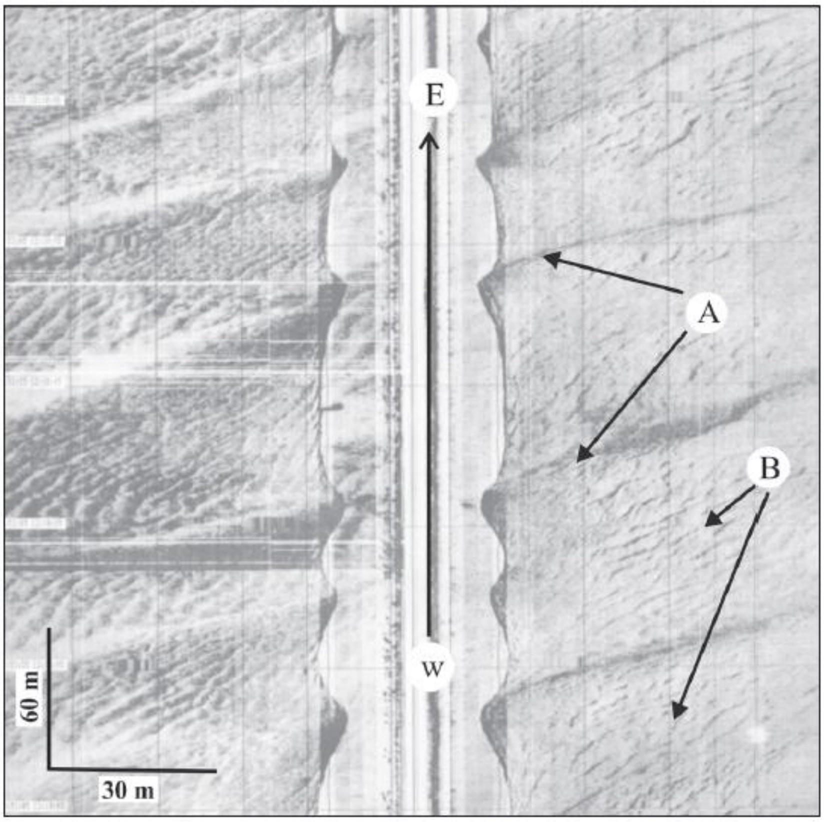


Fig. 4. Sidescan sonar sonograph from the Bristol Channel (UK), showing a field of sand subaqueous dunes (Velegrakis et al., 1996). Key: A, large subaqueous dunes; B, medium subaqueous dunes superimposed on the larger bedforms. Figure taken from Velegrakis et al. (2010).

2.5 BIOLOGICAL RESOURCES IN VICINITY OF EXTRACTION AREAS

The sediments targeted by the aggregates industry are, as a result of low light penetration, predominately characterised by animal communities (Tillin et al., 2011). These animal communities include the macrobenthos, those animals living in close association with the seabed, and more mobile fish and shellfish species. The macrobenthos include infaunal species which live within the sediments, and epifaunal species which live on the sediments. The epifauna comprise of mobile and sedentary species. Many of the sedentary taxa are colonial and require coarse sediment for attachment (e.g. hydroids and bryozoans).

The composition of benthic faunal communities is influenced by the nature and dynamics of the sediments. For example, in stable gravel areas the fauna are typically more diverse and include a well developed epifaunal component (e.g. Cooper, 2013[1]). In contrast, the fauna characterising dynamic sandy areas are typically much less diverse, and are dominated by a relatively small number of highly adapted species (e.g. Cooper et al., 2007a[11]). A combination of a lack of surfaces for attachment, and the harsh nature of the environment results in colonial epifaunal species being less common (Kenny et al., 1991). Evidence from Emu Ltd (2010) suggests that the benthic macrofaunal communities found in association with the relatively deep deposits of aggregate extraction areas are not unique, and can be found associated with more shallow deposits in the near vicinity.

The seabed in the vicinity of aggregate extraction areas can also support a variety of commercially important fish and shellfish species. For example, coarse sediments are utilised by herring (de Groot, 1980) and black bream (Defra, 2002) for spawning, and female brown crabs are known to bury in shingle during an over-wintering phase (Bennett & Brown, 1998). Many demersal species rely on benthic organisms for their diet (Greening and Kenny, 1996; Pearce, 2008).

2.6 ENVIRONMENTAL IMPACTS

The typical direct and indirect effects of aggregate dredging are shown in Figure 5 and are described below. It is difficult to generalise about impacts of dredging, as these will depend on local environmental conditions, and the dredging practices employed at the site (e.g. intensity and method of dredging, use of screening). In some locations, impacts of dredging on the seabed are clearly visible, whilst in other locations impacts are much less obvious.

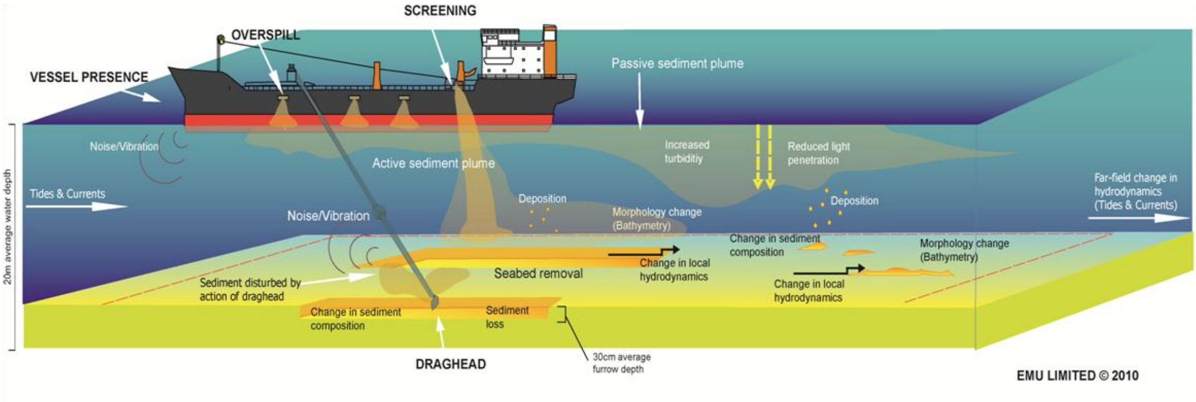


Fig. 5. Illustration of the direct (primary) and indirect (secondary) impacts associated with marine aggregate extraction (Copyright Emu Ltd). Figure taken from Ware & Kenny (2011).

2.6.1 Direct physical effects

For trailer dredging, contact of the draghead with the seabed leads to the creation of dredge furrows (Limpenny et al., 2002) (see Figure 5). These features are typically 2-3m wide and 20-30 cm deep, but can coalesce to form larger bathymetric depressions. The persistence of dredge furrows varies considerably from a matter of days to years, depending on the nature of the local environment (Limpenny et al., 2002). For anchor dredging, contact of the draghead leads to the creation of dredge depressions or pits (see Dickson and Lee, 1972). These features have been reported to be over 20m in depth and up to 75m across (Newell et al., 1998). In most cases, dredge depressions are likely to be long-term, or even permanent, features of the seabed. Changes in the topography of the seabed resulting from furrows and depressions can alter the local hydrodynamics, with possible implication for sedimentation and the trapping of finer sediments (Dickson and Lee, 1972; Kenny et al., 1998; Newell et al., 1998, 2004a). Changes in sediment composition can also occur within the licence area as a result of the exposure of different underlying material (Kenny and Rees, 1996; Cooper et al., 2007b[10]).

2.6.2 Direct biological effects

Benthic organisms are vulnerable to entrainment (direct uptake by the suction field generated at the draghead). For the macrofauna, figures vary for the severity of impact, with reported reductions in abundance of 16% to 88%, in diversity of 11% to 76%, and in biomass of 78% to 92% (see Newell et al., 1998, 2004b, Andrews 2004). Whilst benthic organisms which are taken up by the dredger could, in theory, be returned via overspill or screening, evidence suggests most of them do not survive Newell et al. (1999). Evidence in Poiner and Kennedy (1994) and Newell et al. (1999) suggests this organic material may be associated with an enhancement or 'halo' effect on the benthos at distance away from the extraction site, but such results are inconclusive.

For fish, it is the demersal species which are most vulnerable to entrainment (Drabble, 2012), although pelagic species can also be affected (Reine & Clarke, 1998). Also vulnerable to entrainment is the brown crab *Cancer pagurus*, particularly the egg-carrying females which can be found over-wintering in coarse sediments (Bennett and Brown, 1998). A study by Lees et al. (1992) showed living fish being returned to the water via overspill, although the study did not consider long-term survival. Dredging might also affect fish populations through damage to spawning and feeding grounds, and the entrainment of eggs and larvae (Tillin et al., 2011).

Biological effects are likely to persist in areas of active dredging. However, upon cessation of dredging, faunal recolonisation does occur following a pattern of normal successional changes (Boyd et al., 2005[12]). This recolonisation may result in the return of a similar community to that which existed before dredging, or in a different community, depending on the severity of persistent physical change (Poiner and Kennedy, 1984; Desprez, 2000; Boyd et al., 2005[12]; Barrio Froján et al., 2011[6]; Wan Hussin et al., 2012[4]). Equally, the capacity of the seabed to provide the same functions in terms of habitat for fish may also change depending on the extent and persistence of physical change.

2.6.3 Indirect physical effects

Indirect effects of dredging occur as a result of the mobilisation and deposition of sediment (Figure 5). This happens at the seabed as a result of the disturbance caused by the draghead, and material mobilised in this way is referred to as a benthic plume (Hitchcock and Bell, 2004). Sediment plumes also originate from the dredger. These surface plumes arise from the processes of overspill (Newell et al., 1998), where excess water and fine sediments are lost over the side of the vessel. They also result from the process of screening, where unwanted sediment fractions are returned to the seabed (Hitchcock and Drucker, 1996). The proportion of material lost through screening will vary according to the composition of dredged deposits and the targeted sediment fractions. However, in a study of two sites in the southern North Sea, Newell et al. (2004a) calculated that 57.7% and 28.5% of the material dredged was returned to the seabed as a result of this process. In some extreme circumstances, Hitchcock and Drucker (1996) noted that up to 80% of the material initially removed from the seabed can be returned through screening. According to Hitchcock and Bell (2004), the practice of screening can lead to oversanding, making it increasingly difficult to load coarse cargoes.

The denser components of a sediment plume generally fall to the seabed within 300-600m from the point of discharge (Poiner and Kennedy, 1984; Hitchcock and Drucker, 1996; Newell et al., 1999, 2004a; Hitchcock et al., 1999). However, there is some evidence that benthic plumes can propagate further (Hitchcock and Bell, 2004; Dickson and Rees, 1998). In contrast, a surface plume of the finer sediments can extend for up to 2.5 km from the point of discharge. Whist the settlement of material is relatively localised, effects can extend further due to subsequent remobilisation of settled material on successive tides (Cooper et al., 2007a[11]).

Where the supply of rejected material from a dredger exceeds the capacity of the environment to disperse it then this can lead to changes in the character of the seabed. For example, in the near vicinity of the extraction area, the settling plume material can lead to the creation of sediment bedforms, including sandwaves and sand ripples (Boyd et al., 2005[12]). Further afield, settling material can lead to a change in the composition of seabed sediments (e.g. Desprez and Duhamel, 1993; Desprez, 2000; Newell et al., 2004a; Cooper et al., 2013[2]). The extent to which these changes occur is dependent on a variety of factors including the nature of the receiving environment, and the extent of screening. In some locations, such as the Hastings Shingle Bank (Cooper et al., 2007b[10]), there is relatively little evidence of changes in sediment composition outside the licence area. In other areas changes in sediment composition are more widespread (Newell et al., 2004a).

2.6.4 Indirect biological effects

The indirect physical impacts of dredging described above can lead to impacts on the benthic fauna, although such effects are likely to be less severe than in the primary impact zone. Indirect or secondary impacts result from increases in Suspended Particulate Matter (SPM), and from the settlement of sediment on the seabed (Newell et al., 1998). Newell et al. (2004a) showed evidence of an impact to the macrofauna up to 4000m away from the dredge area. However, secondary impact zones for biota are typically much smaller (e.g. Desprez, 2000; Boyd and Rees, 2003; Newell et al, 2004a; Desprez et al., 2010; Pearce et al., 2011). Increases in SPM as a result of sediment plumes can impact benthic organisms through increased scour (Tillin et al., 2011), or through damage and blockage to feeding and respiratory organs (Newell et al., 1998). For example, Last et al. (2011) showed that scallops exhibited an increase in 'coughing' in response to increases in SPM. This behavioural response is thought to be caused by a need to expel sediment from inside the mantle cavity, and, given the energetic costs it may affect growth. Interestingly, the same authors described a positive effect of intermediate levels of SPM on the growth of *Sabellaria* reef. Last et al. (2011) also considered the effect of burial on a range of benthic organisms. They found a high tolerance of burial for some species (e.g. *Sabellaria spinulosa*), intermediate tolerance for *Mytilus edulis* (the blue mussel) and *Ciona intestinalis* (yellow sea squirt), and low tolerance for *Psammechinus miliaris* (green sea urchin). Changes in sediment composition, both during and after cessation of dredging, can lead to alterations in the composition of the benthic community (Poiner and Kennedy, 1984; Desprez, 2000; Boyd et al., 2003[15], 2005[12]; Barrio Froján et al., 2011[6]).

2.7 LICENSING PROCESS

In the UK, sand and gravel resources on the seabed out to the 200 nm limit are owned by the Crown Estate. In accordance with their responsibilities under the Crown Estate Act (1961), the Crown has a duty to generate an income, on behalf of the exchequer, from the exploitation of this material. This is achieved by the issuing of exclusive production agreements which give commercial dredging companies the right to extract material from within defined extraction areas. In return, the dredging company pays a royalty to the Crown for every tonne of aggregate produced. The awarding of a production agreement by the Crown follows a tendering and prospecting process, and is dependent on the developer successfully obtaining a Marine Licence from the relevant Regulator.

A Marine Licence is required, under the Marine and Coastal Access Act 2009 (MCAA), for activities involving the deposit or removal of a substance or object from below mean high water springs, and this includes the extraction of marine aggregates. Marine Licences are issued by a variety of regulatory organisations depending on country and distance offshore (see Table 1).

Table 1. Regulatory bodies responsible for issuing Marine Licences within UK waters. Abbreviations: MMO - Marine Management Organisation, WG - Welsh Government, DOENI - Department of Environment Northern Ireland, NIEA - Northern Ireland Environment Agency, SG - Scottish Government. The MMO is an executive non-departmental public body (NDPB) established and given powers under the Marine and Coastal Access Act 2009.

Country	0 – 12 nm	12 - 200 nm
England	MMO	MMO
Wales	WG	MMO
Northern Ireland	DOENI/NIEA	MMO
Scotland	SG	SG

Marine Licence applications are considered in accordance with relevant government policy statements including the Marine Policy Statement (UK Government, 2011) and relevant Marine plans and guidance, and principles of sustainable development. In addition, they are also subject to assessment under the Habitats Regulations, the Water Framework Directive (out to 1 nautical mile), and they may be subject to an Environmental Impact Assessment (EIA).

The EIA directive (Council Directive 85/336/EC) aims to protect the environment and the quality of life by ensuring that projects which are likely to have significant environmental effects by virtue of their nature, size or location are subject to an environmental impact assessment before permission is granted. The extraction of minerals by dredging is listed in Annex II of the EIA directive meaning that if the project is likely to have a significant environmental effect then it will require an EIA. In practice all UK aggregate dredging projects have been subject to EIA. The EIA directive was enacted into UK law by the Marine Works (Environmental Impact Assessment) Regulations 2007 as amended in 2011 (MWR).

As part of an EIA, the developer is required to characterise the benthic fauna within the likely area of effect, and then to consider what the impacts of the project are likely to be. If necessary, the developer may suggest some form of mitigation to minimise or avoid impacts on sensitive species or habitats. The results of the EIA are presented in an Environmental Statement which is presented to the regulator, along with an application for a Marine Licence. In making their decision on the Marine Licence application, the regulator will seek comment from statutory consultees, and their technical scientific advisors. To mitigate the effects of dredging, conditions are often applied to extraction licences. Examples of licence conditions include: (1) limits on the extraction rate, (2) limits on the total tonnage extracted, (3) restrictions regarding the quantity of material which can be screened, (4) a requirement to leave the seabed in a similar physical condition after dredging, and (5) a requirement to monitor the environmental effects of dredging over the licence term (see Ware and Kenny, 2011).

3. MY RESEARCH

Findings from the research questions I have addressed over the last 11 years have informed the rationale for a new approach to monitoring the impacts of marine aggregate dredging (see Section 5). These research questions relate to themes of **Impacts (Cumulative Effects)**, the **Relationship between Sediments and Benthos**, **Recovery**, **Restoration**, **Seabed Mapping**, and **Natural Variability**. In this section I undertake a critical examination of this work in order to provide confidence in the new monitoring approach.

3.1 IMPACTS (CUMULATIVE EFFECTS)

An understanding of the nature and spatial scale of dredging effects is important if we are to avoid unacceptable impacts on other legitimate marine interests. Whilst impacts of dredging on benthos and sediments are reasonably well understood at individual extraction sites (Newell et al., 1998), much less is known about the potential for cumulative impacts that may arise from combinations of licensed areas in close proximity, for example in the East Coast dredging region (see Figure 2). As part of the Environmental Impact Assessment process, developers of extraction sites are required to consider the potential for cumulative and in-combination effects. Cumulative impacts are defined as those which arise from multiple marine aggregate extraction activities within a region. In-combination impacts are the total impacts of all anthropogenic activities within the same region (TEDA, 2010). In the past, cumulative and in-combination assessments have not been satisfactorily addressed due to a lack of suitable regional datasets (licence areas are typically monitored in isolation), and an appropriate assessment methodology.

To address the above issues, research was undertaken to develop a method to assess cumulative effects, and to look for evidence of a broadscale impact of dredging on sediments and benthos in the East Coast dredging region (Cooper et al. 2007a[11]). This study used macrofaunal and sediment data from a broadscale survey carried out across the region in 1998. Samples were assigned to treatment groups based on their position relative to predicted 'direct' and 'indirect' effects of dredging. The zone of 'direct' dredging effects was based on the cumulative dredging footprint indicated by Electronic Monitoring System (EMS) data (for an explanation of EMS data see Cooper et al., 2007a[11],b[10]). The zone of 'indirect' effect was based on a modelled area for secondary effects.

Results in Cooper et al. (2007a)[11] provide evidence of a near-field effect of dredging, with proportionally less gravel and more coarse sand within the 'direct' impact zone. In addition, samples from the 'direct' impact zone had significantly (p<0.05) lower numbers of species ($\bar{x}_{(direct)}$ = 5.5,

$\bar{x}_{(reference)}$ = 18.9) and individuals ($\bar{x}_{(direct)}$ = 11.4, $\bar{x}_{(reference)}$ = 287.1) relative to the reference zone. Therefore, we rejected the null hypothesis that there was no difference in the number of species and individuals between the different treatment groups. Further, Cohen's effect size values suggested a large practical significance for the number of species (d = 0.92) and a moderate practical significance for the number of individuals (d = 0.63). The study also showed some evidence for a far-field effect of dredging within the 'indirect' impact zone, with intermediate numbers of species and individuals relative to the 'direct' and 'reference' treatments, but a lack of statistical significance meant that no firm conclusions could be drawn.

The lack of baseline data for the East Coast dredging region means that we could not exclude the possibility that the observed differences between the 'direct', 'indirect' and 'reference' treatments were unconnected with dredging. In addition, the failure to detect a statistically significant difference in the number of species and individuals between the 'indirect' and 'reference' treatment may have been a result of the low numbers of samples available.

Despite the limitations of the approach outlined in Cooper et al. (2007a)[11], it has found application in follow-on work. For example, it is now being used for assessing cumulative effects of dredging in a number of the major dredging regions (e.g. EMU Ltd, 2012). The greater number of samples available to these studies allowed data to be stratified according to the underlying habitat or biotope, providing some insight into potential differences in biotope/habitat sensitivity to dredging.

The identification of an historic footprint of dredging effect, following the approach outlined in Cooper et al. (2007a)[11], was also used to help industry transition to a new system of marine licensing. This issue arose due to the introduction of the Marine and Coastal Access Act (UK MCAA, 2009) which required developers to obtain a Marine Licence for all extraction areas. The time required to complete an EIA for a typical 15 year Marine Licence meant that dredging operations would have had to stop whilst the work was completed, with implications for continuity of aggregates supply and industry jobs. To address this, developers were issued with short-term Marine Licences which restricted their operations to within the historic footprint of dredging. This strategy lowered the level of environmental risk, making the EIA for the short-term Marine Licence much quicker to undertake. As a result, dredging operations were able to continue. During the period covered by the short-term Marine Licence, developers were able to produce an EIA for a full 15 year dredge term.

In addition to the ability to assess for cumulative effects, the regional approach to assessing environmental impacts of dredging on the seabed offers a range of other potential benefits. For example, the collected data allow for a snap-shot of the regional status of the seabed macrofaunal communities. There are also benefits in terms of efficiency, with samples collected from across the region in one sampling campaign. Finally, reference sites can be shared between extraction areas - this is particularly useful where such sites are difficult to find due to the proximity of other activities.

3.2 RELATIONSHIP BETWEEN SEDIMENTS AND BENTHOS

In a review of the relationship between sediments and benthos, Gray and Elliott (2009), highlight the important role of hydrodynamics in influencing the composition of sediments. For example, muds generally occur in low energy, depositional areas, whilst gravels tend to occur in higher energy, more erosive areas. The composition of sediments affects seabed stability and other properties such as porosity and permeability and ultimately sediment chemistry. Together, these variables influence the composition of the benthos. As the aggregates industry target relatively clean (i.e. low silt/clay content) sands and gravels, variables such as static particle size and mobility of sediments are likely to be important factors influencing the biology.

It is generally accepted that the composition of sediments plays an important role, at large spatial scales, for the structuring of benthic faunal communities (Gray, 1974; Warwick and Davies, 1977; Barry and Dayton, 1991; Petersen, 1913; Glémarec, 1973; Dankers and Beukema, 1981; Künitzer et al., 1992). It is for this reason that operators of marine aggregate dredging sites are required, through a licence condition, to leave the seabed in a similar state to that which existed before dredging (ODPM, 2002). The rationale for this condition is to promote recovery, and the return of a similar faunal community to that which was present before dredging. Careful stewardship of the environment in this way is intended to help avoid long-term cumulative effects, thus helping to ensure the sustainability of aggregate dredging.

At smaller spatial scales, the relationship between sediments and macrofauna can be more variable. For example, Hitchcock and Bell (2004), working off the south coast of the UK, found that a benthic community adjacent to dredging was unaffected by a small change in the composition of sediments. Similarly, Boyd et al. (2005)[12], in a study of recovery following marine aggregate dredging at Area 222 (outer Thames Estuary, UK), found a complete faunal recovery in an area where a small physical impact persisted. Working at extraction sites in the southern North Sea, Seiderer and Newell (1999), Newell et al. (2001) and Cooper et al. (2007a)[11] all reported a lack of a close correspondence between the composition of sediments and the distribution of benthic faunal communities. These studies contrast to results in Desprez (2010) who observed a strong correlation between sediment composition and benthic fauna.

The purpose of the Cooper et al. (2011b)[5] study was to investigate the sensitivity of faunal communities to changes in sediment particle size composition, and to identify the factors which may be responsible. This understanding was intended to help regulators decide where they should, and, perhaps, should not be concerned about changes in sediment composition as a result of dredging. Results from this study showed that faunal communities in stable, gravel rich areas are most sensitive to changes in sediment particle size composition, with faunal communities in dynamic, sandy areas being least sensitive to sediment changes. It is suggested that these findings may, in part, be explained by the close association of certain taxa with the gravel fraction, and the influence of natural physical disturbance which, as it increases, tends to restrict the colonisation by these species. The findings of this study are in agreement with Seiderer and Newell (1999), Newell et al. (2001) and Cooper et al. (2007a)[11], who all reported a lack of correspondence between community composition of the benthos and static particle size distribution in unconsolidated sand and gravel deposits at Area 452 (Thames), off Folkestone (eastern English Channel) and Cross Sands (East coast), respectively. All of these areas were shown by Cooper et al. (2011b)[5] to fall within zones of high natural physical disturbance. The differing sensitivity of faunal communities to changes in sediment particle size composition helps explain the different physical and biological recovery times following marine aggregate dredging reported in Foden et al. (2009).

The findings from Cooper et al. (2011b)[5] suggest that it may be preferable, where a choice exists, to site extraction areas in more dynamic environments, due to the typically lower sensitivity of the benthic communities present within these areas. The study also suggests that there should be more concern about changes in sediment composition in stable gravel areas, and less concern in dynamic sandy areas. Whilst this study improved our understanding of the sensitivity of faunal assemblages to changes in sediment composition, there was still a need for the setting of quantifiable limits for acceptable change in sediment composition to allow regulators to effectively manage the industry (Cooper et al., 2012[3], 2013[1]).

3.3 RECOVERY

The sustainability of marine aggregate dredging for seabed faunal communities is ultimately dependent on the extent of seabed recovery post dredging. Previously, faunal recovery times were expected to be in the range of 2-4 years (Kenny et al., 1998; Sardá et al., 2000; Van Dalfsen et al., 2000; Van Dalfsen and Essink, 2001). However, as these studies were associated with short dredging campaigns (typically of <1 year), there was a concern the results might not be relevant to commercial extraction sites where dredging is sustained over a much longer period, typically 25 years. To address this issue, the UK government commissioned Cefas to undertake research to look at recovery times at commercial extraction site around the UK (see Boyd et al., 2003[15], 2005[12]; Cooper et al., 2007b[10]; Barrio Froján et al., 2011[6]; Wan Hussin et al., 2012[4]).

The research undertaken by Cefas investigated the physical and biological recovery of the seabed at 3 commercial extraction sites where dredging had been sustained over many years. These sites differed in the time since dredging had ceased, and so the study made it possible to look at recovery at different stages of the process. In addition, sites were located in three different regions (Humber, Thames and Eastern Channel), and were subject to different environmental conditions and dredging practices. All of the studies used the same experimental design, with grab samples taken within areas of relatively high and low dredging intensity, and at two reference sites. This made it possible to investigate the effect of dredging intensity on recovery times. In addition to the grab samples, sites were monitored using both acoustic and underwater video, making it possible to monitor the recovery of physical impacts, such as dredge tracks. In these studies biological recovery was defined as the establishment of a community that was virtually indistinguishable from surrounding, non-impacted reference sites. Differences were assessed both uni- and multi-variate analysis techniques.

Findings from this research suggest that faunal recovery can be strongly linked to physical recovery. For example, four years after cessation of dredging at Area 408, there was still no evidence of faunal recovery within the areas of high and low dredging intensity (Barrio Froján et al., 2011[6]). In both these areas, sediments remained very different to the references sites. In contrast, a full faunal recovery was seen within the low dredging intensity areas of Area X (Cooper et al., 2007b[10]), and Area 222 (Wan Hussin et al., 2012[4]) some 6-7 years after cessation of dredging. The sediments within both these areas were similar to the reference sites. Whilst there was progress towards recovery at the high dredging intensity site at Area 222, the persistent physical impacts observed in this location appear, in part, to be responsible for the prolonged faunal recovery. According to Wan Hussin et al. (2012)[4], biological recovery at the high site is expected to take between 15-20 years. The results of the Cefas recovery studies have been put into a wider context in Foden et al. (2009). Whilst there is still much variability in reported recovery times, Foden et al. (2009) shows the fastest recovery rates occur in dynamic areas with fine sediments (e.g. mobile sands), and the longest times in stable coarse sediment areas. Other authors have approached the recovery question in a different way by looking at the composition of benthic communities and predicting how long they might take to re-establish based on the life-history traits of the species present (MESL, 2007). Whilst this approach is potentially useful, no specific recovery time predictions are made. In addition, the report highlights a need for testing predicted recovery times against data from field-based recovery studies. The approach also assumes that the physical habitat, post dredging, is able to support the return of the pre-dredge faunal assemblage. Work in Wan Hussin et al. (2012)[4] shows this assumption may not always be valid.

In addition to assessing recovery in terms of community structure, attempts have also been made to look at recovery in terms of likely ecosystem function (Cooper et al., 2008[8]; Barrio Froján et al., 2011[6], Wan Hussin et al., 2012[4]). Ecosystem functions performed by the benthos include metabolism, catabolism, bioturbation, production and transfer of food, oxygen, and nutrients and recycling of waste and harmful substances. Whilst we were not able to measure these processes directly, we sought to gain an insight into possible differences in ecosystem function using a variety of different functional metrics (e.g. Infaunal Trophic Index (ITI), Somatic Production (P_s), Biological Traits Analysis (BTA), Taxonomic Distinction (TD) and Rao's Quadratic Entophy coefficient (Rao's Q) (see papers listed above for further details of each method). This different approach to assessing faunal recovery was trialled to assess the likely ecological significance of changes in benthic community composition. The logic of this is that whilst the structure of a recolonising community may differ to that present originally, or to that found at a reference site, it may never-the-less perform some or all of the same ecosystem functions. Assessing recovery in this way can therefore provide a greater insight into the likely significance of environmental change. Results of this work suggested that the impacted seabed might recover its functionality in advance of the recovery times suggested by the species abundance data. However, large differences in community structure (species composition) were also associated with large differences in ecosystem function, as suggested by a comparison of biological trait expression (Barrio Froján et al., 2011[6]).

The most obvious criticism of the Cefas recovery studies is the lack of baseline data. As such, we cannot be certain that the differences between dredged and reference sites are entirely down to dredging. This issue was, as far as possible, addressed by carefully selecting reference sites which would be representative of the pre-dredge state. A further criticism relates to the possible confounding influence of other activities across the site (e.g. demersal fishing). This is a difficult issue to address, but access to Vessel Monitoring System (VMS) data, which shows the location of fishing vessels (Eastwood et al., 2007), could now help. In the work undertaken to assess functional recovery (Cooper et al., 2008[8]; Barrio Froján et al., 2011[6], Wan Hussin et al., 2012[4]), no attempt was made to link specific biological traits to specific ecosystem functions. This is a weakness which more recent studies have sought to address by first identifying the functions of interest, and then selecting only the biological traits of relevance for subsequent analysis (see Frid, 2011).

Work on recovery has shown that it is possible to get a return of the original faunal assemblage, given time, where the physical environment remains within certain limits (Cooper et al., 2007b[10]; Boyd et al., 2005[12]; Wan Hussin et al., 2012[4]). The work of Cooper et al. (2011b)[5] suggests that the consequences of physical changes in seabed condition for faunal recovery are likely to vary according to the sensitivity of the affected faunal assemblage.

3.4 RESTORATION

Concerns about the limited recovery potential of some dredge sites (Desprez, 2000; Boyd et al., 2005[12]; Barrio Froján et al., 2011[6]; Wan Hussin et al., 2012[4]), combined with various legislative and policy drivers (e.g. Article 2 of the EEC Habitats Directive (Council Directive 92/43/EEC), the EC Water Framework Directive (Directive 2000/60/EC), the EC Marine Strategy Framework Directive (Directive 2008/56/EC), Article 2 of Annex V to the OSPAR Convention (OSPAR 1992) and the Environment Liability Directive (Directive 2004/35/EC)), led to a consideration of possible restorative techniques. A review of these techniques can be found in Cooper et al. (2013)[2]. Whilst many of these techniques are routinely used for maintenance dredging, it is important to acknowledge that their validity for restoration of aggregate sites remains largely untested. There are a couple of exceptions to this. Firstly, the work of Collins and Mallinson (2006) which considers the use of waste shell material, and a gravel seeding experiment undertaken by Cooper et al. (2011b)[7]. The objective of the Cooper et al. (2011b)[7] study was to assess the feasibility and effectiveness of gravel seeding as a means of recreating a lost coarse sediment seabed habitat (see Barrio Froján et al., 2011[6]). The survey design comprised of three boxes. Two of the boxes were located within an area of sandy sediments thought to have resulted from screening of dredged cargoes. The other box was located in a reference area characterised by gravel sediments. Within the sandy area, one of the boxes was designated as the treatment site, the other as a control. The treatment site was seeded with a thin layer of gravel rich sediments, using a commercial aggregate dredger. Over time the faunal composition of the treatment site was expected to become more similar to that of the gravel reference site.

Results from this study showed it was possible to increase the proportion of coarse sediment at the seabed surface within the treatment box, although it demonstrated the difficulty in trying to exactly recreate a lost habitat. The study also showed that the technique was successful, at least in the short-term, in returning the benthic community to a state more similar to the surrounding gravel sediments in the wider region. Knowledge of the fauna in the vicinity of the seeding area (Barrio Froján et al., 2011[6]) lends support to the existence of the treatment effect, but a lack of replication in the survey design means we need to be cautious about a generalised effect of gravel seeding elsewhere. Unfortunately, overriding practical considerations prevented an ideal survey strategy. Had resources been available, the design could have been improved by replicating the treatment and reference sites, and by monitoring the site on multiple occasions before and after dredging (Hurlbert, 1984). These changes would have resulted in a true, Before-After Control-Impact (BACI) approach, allowing for more robust conclusions concerning the existence of a generalised treatment effect.

The considerable effort required to restore the sediment composition within a relatively small part of extraction Area 408 illustrates the inevitable costs of restoration. The issue of cost, financial and environmental, is considered in Cooper et al. (2013)[2]. This study investigated whether restoration could be justified using a good and services valuation approach, but the lack of suitable data meant that this was presently not possible, other than in a qualitative sense. The study therefore recommended that there was a need for a more objective way of deciding whether restoration is necessary. In the longer term, the approach used in Cooper et al. (2013)[2] could be useful for making decisions about whether it is appropriate to restore the seabed where acceptable change limits are breached. However, for this to happen it would require valuation data to be collected at a much finer resolution than at present. The issue of who should pay for restoration was also addressed by Cooper et al. (2013)[2]. They concluded that a centralised fund might be most appropriate as it would remove the financial burden from any one developer. An example of such a fund was the now discontinued Aggregate Levy Sustainability Fund (ALSF), which aimed to reduce the environmental impacts of the extraction of aggregates and to deliver benefits to areas subject to these impacts (Defra, 2010).

3.5 HABITAT MAPPING

The effective management of marine resources requires a thorough knowledge of their distribution. In the context of marine aggregate dredging, it is very difficult to judge the significance of changes to benthic faunal communities without knowledge of the extent of their distribution in the wider region. For this reason, considerable effort has recently gone into producing habitat/biotope maps in regions of marine aggregate dredging (e.g. Mackie et al., 2006; James et al., 2007; Emu Ltd, 2009; James et al., 2010; Limpenny et al., 2011; Tappin et al., 2011; James et al., 2011). The mapping approach used in these studies was initially developed in the UK by Brown et al. (2002[16], 2004a[13],b[14]). Their work showed seabed acoustic data (e.g. sidescan sonar, acoustic ground discrimination system (AGDS), and multibeam bathymetry), in conjunction with ground-truth grab (for fauna and sediments) and camera samples, could be used to construct habitat/biotope maps.

The benthic and sediment data generated by the broadscale mapping initiatives are key to identifying limits for acceptable change in sediment composition within the footprint of dredging effect (Cooper, 2012[3], 2013[1]). Rather than using the biotopes identified in the broadscale mapping initiatives, the method in Cooper (2012)[3] and Cooper (2013)[1] uses the macrofaunal data to identify what broadscale communities are present within the region. The range of sediment composition found in association with each of these faunal groups is then used to define limits for acceptable change. The logic being that as long as sediments remain within this range then it should be possible for the original faunal assemblage to return after dredging.

3.6 NATURAL VARIABILITY

Long-term records of biological data are extremely valuable for documenting ecosystem changes, for differentiating natural changes from those caused by humans, and for generating and analyzing testable hypotheses (Wolfe et al., 1987). To address the lack of long-term datasets for marine aggregate dredging, Cefas started a network of Environmental Assessment Reference Stations (EARS) in the East Coast dredging region (Barrio Froján et al., 2008[9]). The stations sampled as part of this study were a subset of those sampled during an earlier broadscale survey (Cooper et al., 2007a[11]). The aim of the time-series investigation was to determine whether the broadscale spatial pattern of sediments and benthos observed by Cooper et al. (2007a)[11] was maintained over time. Results showed that this was the case, and the persistence of the pattern over an eight year period lends support to the existence of a dredging effect with the licence areas. The study also noted that sediments and benthos within the area subject to dredging were more homogenous and faunally impoverished than those outside. This reduced variability is attributed to the frequent disturbance associated with dredging, which is thought to dampen the existing naturally high variability in assemblage structure by removing and/or preventing the reestablishment of a more mature and diverse benthic assemblage. The temporal data also showed that undesirable changes did not persist, suggesting that the faunal community was resilient to change. Barrio Froján et al. (2008)[9] also show how an understanding of natural variability can be used to identify when conditions go beyond what might be expected naturally. This kind of information is vital to allow managers to know when to intervene.

4. CURRENT MONITORING APPROACH

4.1 DESCRIPTION

At present, there are two types of benthic surveys undertaken in support of marine aggregate dredging. These are the characterisation survey which informs the EIA, and monitoring surveys which are undertaken periodically once dredging has begun. A detailed description of the survey rationale for characterisation and monitoring surveys is given in Ware & Kenny (2011), but the main features of each survey are provided below.

4.1.1 Characterisation survey

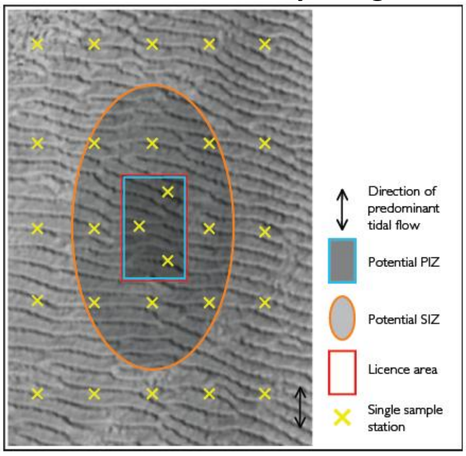
The purpose of the characterisation survey is to provide a description of the sediments and fauna within the footprint of likely dredging effect. The survey should also extend outside this zone to allow for the identification of suitable future reference sites for monitoring. To maximise the spatial coverage of the survey, single samples are acquired from each station. Where available, information on the underlying sediment strata can be used to inform the survey design. For example, where the underlying substratum is homogenous, or the nature of the seabed is unknown, then a regular grid strategy is used to position samples (Figure 6a). For a heterogeneous seabed, samples are typically randomly stratified according to the underlying strata (Figure 6b). The need to undertake a characterisation survey will be determined by the availability of existing data. In recent years, the completion of a number of regional broadscale seabed characterisation studies (e.g. James et al., 2007, 2010; EMU Ltd, 2009; Limpenny et al., 2011; Tappin et al., 2011) has reduced the need for additional site-specific characterisation studies. The results of the characterisation data are used in the EIA to inform what the likely impact of the project will be on the faunal communities present. Where sensitive or protected species occur then appropriate mitigation can be put in place, or in an extreme case, the project may be abandoned altogether. The role of the regulator, with input from their scientific advisors and statutory consultees, is to indicate whether they agree with the developer's interpretation of the data. Where a marine licence is given, then the results of the characterisation survey will be used to inform the design of the subsequent seabed monitoring programme.

4.1.2 Monitoring surveys

The aim of monitoring is to demonstrate that impacts to the seabed are no greater than those predicted in the Environmental Statement. Where impacts are shown to be greater than predicted, the regulator may revise the licence conditions, or, in an extreme case, revoke the licence altogether. The recommended monitoring survey design differs according to whether the seabed is homogenous (Figure 6c) or heterogeneous (Figure 6d). Samples are stratified according to both the underlying sediment strata, and the predicted zone of impact. The predicted impact zones are the Primary Impact Zone (PIZ), the Secondary Impact Zone (SIZ) and the Reference zone (REF). These zones correspond to where the direct, indirect and no impacts of dredging are expected to occur. Within each sediment/impact zone, a number of randomly position $0.1m^2$ grab samples are acquired from within a sampling box. These samples are considered to be replicates, and a comparison of data resulting from the PIZ, SIZ and REF of each sediment stratum allows for an assessment of dredging effect. Monitoring surveys are undertaken for a pre-dredge baseline, and then at regular intervals, the frequency of which is determined by the perceived sensitivity of the environment. The last survey takes place shortly after the end of the licence term, making it difficult to address questions regarding recoverability of the site. Where there are multiple biotopes present within an area then it is likely that only the most sensitive will be monitored.

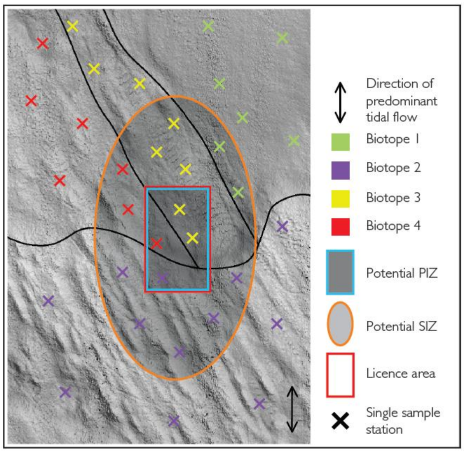
Characterisation survey designs

a)



Homogenous seabed/unknown strata

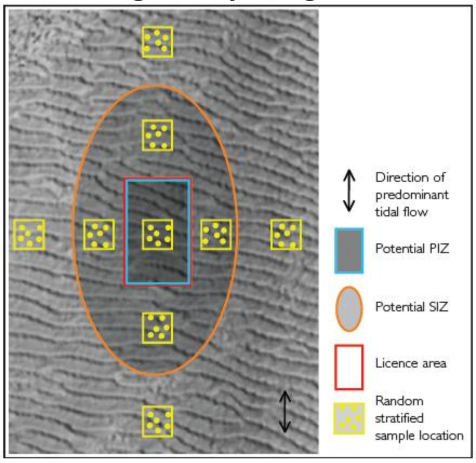
b)



Heterogeneous seabed

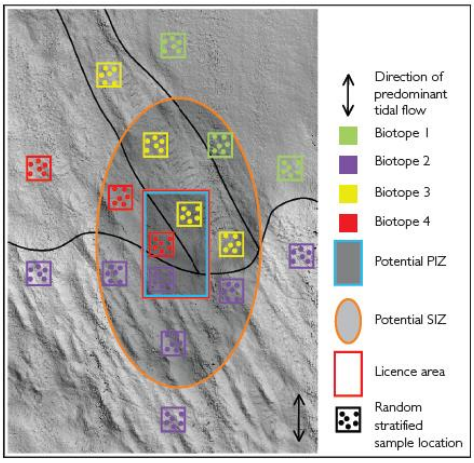
Monitoring survey designs

c)



Homogenous seabed

d)



Heterogeneous seabed

Fig. 6. Recommended designs for characterisation and monitoring surveys under the current approach to monitoring (Figure adapted from Ware and Kenny, 2011). No spatial scale is provided in this figure as the size of extraction sites (PIZ) and their secondary impact zones (SIZ) are highly variable (see Figure 2 and Sections 2.6.3 and 2.6.4 for further details).

4.2 PROBLEMS

4.2.1 Wrong question

The current monitoring approach seeks to answer the question *'What is the impact of ongoing dredging on sediments and fauna?'*, and to confirm that the observed impacts are no greater than what was predicted in the Environmental Statement. There are two reasons why I believe this is now the wrong question:

1. The increasing number of impact studies means that dredging effects are, to some extent, predictable; the current monitoring approach simply confirms what we might expect.

2. Most importantly, the results of current monitoring efforts do not adequately inform us what is likely to happen to the site post-dredging (i.e. recoverability). At present our understanding of seabed recovery at commercially exploited extraction sites is based on only a small number of case studies (see Foden et al., 2009).

If dredging is to be sustainable, and this is a stated aim of the dredging industry and government, then a more appropriate question would be *'Will the seabed within the footprint of dredging effect be able to support the return of the original faunal assemblage after dredging?'* Answering this question allows the industry and managers to respond to changes, as they happen, with a view to maintaining the seabed environment in an acceptable condition, thus ensuring the long-term sustainability of aggregate dredging on seabed macofaunal communities.

4.2.2 Licence condition difficult to enforce

Developers are currently required by a licence condition to leave the seabed in a similar physical condition to that which existed before dredging (ODPM, 2002). The aim of this condition is to try to ensure the return of a similar faunal community post dredging. The difficulty here is deciding what is and is not similar, and for this reason the licence condition is extremely difficult to enforce. In my view, 'similar' should not mean no change, but that the extent of change should not preclude the return of the original faunal assemblage identified during the regional baseline assessment (see Cooper, 2013[1]). At present, the acceptability of changes in seabed condition relies on a subjective 'expert judgement' from the regulator, and this has implications for the consistency between different licensed areas.

4.2.3 Survey design

The recommended monitoring survey design can also be problematic given the assumptions which need to be made concerning the most likely path and location of dredging impacts. In one case, to the east of the Isle of Wight, grab sampling sites did not, despite best intentions, intersect with the secondary effects seen in the acoustic data of the seabed (see MESL, 2005 and MESL, 2010). Also, for many extraction areas, dredging often takes place within relatively small active zones. Any movement of dredging from one zone to another may make the survey design obsolete. In theory, sampling boxes could be repositioned, but then there is no possibility of acquiring pre-dredge baseline data. It is worth highlighting that a very similar survey design to the one recommended for monitoring (Ware and Kenny, 2011) has been successfully employed in studies concerning recovery (Cooper at al., 2007b[10]; Wan Hussin et al., 2012[4], Barrio Froján et al., 2011[6]) and restoration (Cooper et al., 2011a[7]) of the seabed. However, in these studies, no assumptions had to be made concerning to location of boxes – they were positioned to coincide with past impacts, or themselves subject to a treatment.

4.2.4 Little scope for regional assessment of data

In the Eastern English Channel, monitoring of the ten extraction sites is done as part of a regional programme. This regional approach has many advantages. For example: (i) it offers an efficient way of surveying, (ii) impacts at individual sites can be viewed in the context of the status of the overall region, (iii) data from individual sites can be combined to perform powerful meta-analyses (Cooper, 2013[1]), and to assess for cumulative (Cooper at al., 2007a[11]) and in-combination effects. In all other dredging regions, monitoring of extraction sites is done in isolation (i.e. site specific surveys and reporting), with none of the advantages of the regional approach.

4.2.5 Cost

Whilst the motivation for developing a new approach to monitoring was simply to improve environmental protection, the new approach is likely to have a secondary benefit in terms of a major reduction in the complexity and cost of monitoring. This is useful given the current government's policy of seeking to reduce the regulatory burden on industry (UK Government, July 2013). Costs associated with seabed monitoring are a major expense for the aggregates industry. These costs are expected to rise sharply over the next few years as the number of sites which require monitoring goes up from around 23 to 70. This change results from the introduction of a new licensing regime under the Marine and Coastal Access Act (UK MCAA, 2009). The increasing number of monitored sites will also have implications for the regulator, who will need to find additional resource to process the increasing number of monitoring reports. In addition, the current approach to monitoring sites individually is very inefficient in terms of vessel time. To illustrate this point it is worth comparing the costs of ship time to monitor 10 sites individually, versus the costs to monitor the same sites using one survey campaign. Surveyed individually, each site requires 3 days of vessel time (one day each for mobilisation, survey, and demobilisation). The cost of this approach for all 10 sites would be £30,000, assuming vessel costs of £1000/day. To survey all sites together would require 12 days (1 day each for mobilisation and demobilisation, and 10 days for the survey work). The ship costs of this regional approach would be £12,000.

4.2.6 Lack of confidence in results

The final issue with the current approach concerns confidence in the results of benthic surveys. This issue arises due to the often highly variable nature of benthic faunal assemblages in space and time (e.g. Villnäs and Norkko, 2011). As a result of this variability, the confident detection of change relies on there being sufficient numbers of sample replicates within treatments, and an adequate number of sampling events. Both these issues (replication and the frequency of monitoring) have obvious financial implications for the developer, and hence there is often a tension between what's required for scientific robustness, and what the developer is willing or able to pay for. Whilst the use of statistical power analysis has, more recently, been encouraged (Ware & Kenny, 2011), many benthic monitoring programmes have not explicitly addressed this issue. Possible reasons for this include:

(i) A lack of awareness of statistical power analysis, and how it can be used to identify the required number of samples;

(ii) A fear of what it might tell us. For example, Somerfield (in Cooper et al., 2011) showed that for the reliable detection of a 25% difference in the number of species, some 20 to 100 replicates per treatment would be required, depending on the number of species present in the reference or pre-dredge community.

(iii) The Impact hypothesis set out in an Environmental Statement typically won't specify what changes in benthic community composition are expected, other than in general terms. This may be because such predictions would involve a degree of conjecture, but also because making such a prediction would require the developer to put in place a benthic survey capable of detecting it. Given the likely costs of this approach, predictions have been less explicit, and monitoring has been of a 'reassurance' nature, which seeks to demonstrate that no major change has occurred.

Whilst the regulator has effectively sanctioned the 'reassurance monitoring' approach, it has meant that we don't fully understand the nature of impacts at many extraction sites. Perhaps inevitably, failure to get robust answers leads to further monitoring.

5. NEW MONITORING APPROACH

5.1 DESCRIPTION

My proposed new approach to monitoring environmental impacts of dredging on the seabed is for the establishment of a **Regional Seabed Monitoring Plan (RSMP)** for each major aggregate dredging region (Humber, East Coast, Thames, Eastern English Channel, South Coast, Bristol Channel, North West). The RSMPs would include 3 elements:

1. acceptable change limits for sediment particle size within the footprint of dredging (primary and secondary impact zones),
2. a network of long-term benthic monitoring stations, and
3. dedicated research sites.

The RSMP approach is based on the work described in Cooper (2012)[3], Cooper (2013)[1] and Barrio Froján et al. (2008)[9] and the best practice shown by industry in the Eastern English Channel (EEC). In most cases, the RSMPs would fulfil the site specific monitoring requirements for benthic ecology. However, in a small number of localities, there may still be a need for additional work to address site specific benthic issues (e.g. *Ophiothrix* monitoring at Area 461 in the EEC). Each of the RSMP elements is discussed below.

1. Acceptable change limits for sediment particle size

In the United Kingdom, companies extracting marine aggregate from the seabed are typically required, through a condition attached to the extraction licence, to leave the seabed in a similar physical condition after the cessation of dredging. This requirement, articulated in the government policy document covering marine aggregate dredging (ODPM, 2002), is intended to promote recovery, and the return of a similar faunal community to that which was present before dredging, thus reducing the likelihood of long-term and potentially cumulative impacts on the wider ecosystem.

Evidence suggests this policy is sensible. For example, numerous studies (Desprez, 2000; Newell et al., 2004a,b; Boyd et al., 2005[12]; Robinson et al., 2005; Desprez et al., 2010; Cooper et al., 2011b[5]; Barrio Froján et al., 2011[6]; Wan Hussin et al., 2012[4]) have shown that a change in sediment composition has the potential to alter the benthic community composition, and the potential for recovery. In such cases it is difficult to argue that seabed sediments have remained in a 'similar' condition post dredging. Whilst this policy is clearly sensible, and in line with the principles of sustainable development, the ambiguity associated with the term 'similar' can be problematic for both developer and regulator. For example, developers have no clear definition of what is acceptable or not in terms of changes in sediment composition, and the regulator is forced to make subjective assessments as to the acceptability of changes that may occur. If the condition is to achieve its purpose of mitigating adverse environmental impacts then it has to be enforceable, and this requires it to be specific and measurable. The development of a more objective method of assessment is timely given that approximately forty licence applications for marine aggregate extraction are expected in the next 1–2 years; this is a result of many existing licences reaching the end of their previous terms, and a new licensing system resulting from the Marine and Coastal Access Act (UK MCAA, 2009).

A possible solution to this problem comes from recent Regional Environmental Characterisation (REC) initiatives which have mapped the biological resources present within, and surrounding areas of marine aggregate dredging. The first of these surveys was undertaken in 2005/6 in the Eastern English Channel (James et al., 2007). Subsequent surveys have occurred off the South Coast (James et al., 2010), Thames (Emu Ltd., 2009), East Coast (Limpenny et al., 2011) and Humber regions (Tappin et al., 2011). These surveys, and more localised habitat mapping initiatives (e.g. Boyd et al., 2004; Brown et al., 2004a[13],b[14]; Birchenough et al., 2010; ECA and EMU Ltd., 2010; ERM, 2010), have provided a new understanding of the distribution of benthic faunal communities in regions of marine aggregate dredging. The spatial coverage of the REC surveys has been further enhanced by the aggregates industry through their Regional Environmental Assessment (REA) initiatives which have collected additional data in and around the licence areas within the Humber, East coast, Thames, Eastern English Channel and South Coast dredging regions (MAREA, no date). In addition, and crucially, the collected data make it possible to identify the range of sediment conditions found in association with individual benthic faunal assemblages typical of marine aggregate producing regions; it is this information which has the potential to define limits for acceptable environmental change. In theory, as long as the composition of sediment within an impacted area remains within an acceptable range, as defined by the initial pre-dredge state and comparable conditions in the wider region, then a return of an acceptable benthic assemblage should be possible following the cessation of dredging. Such an approach fits well with results reported by Cooper et al. (2011b)[5], which showed that the sensitivity of benthic faunal assemblages to changes in sediment composition caused by marine aggregate dredging can vary. The suggested approach would allow for this, providing an appropriate level of localised protection. The acceptable change limits would be identified by the regulator and set out in a proposed licence condition during the Environmental Impact Assessment phase of the development. Having established the condition for acceptable change in sediment composition, this would become a focus for the developer lead monitoring, and final post-dredge assessment of seabed status.

The above approach has been tested in two locations, at an extraction site off the south coast at Hastings (Cooper, 2012[3]), and in the Eastern English Channel dredging region (Cooper, 2013[1]). In both localities the approach showed promise, and there is now a consensus amongst stakeholders (Marine Management Organisation, Department for Environment, Food and Rural Affairs, British Marine Aggregate Producers Association, The Crown Estate, Natural England, Joint Nature Conservation Committee, Welsh Government and Natural Resources Wales) that it should be considered for use more widely. For this to happen, acceptable change limits for sediment composition will need to be identified for all the major aggregate producing regions.

2. Long-term benthic monitoring stations

Within each region, a network of benthic monitoring stations (sediment and fauna) will be established in areas outside the impact of dredging. These stations will serve a number of purposes:

i) To allow the broadscale seabed characterisation to be kept up-to-date, reducing the need for additional characterisation surveys in support of new licence area applications.
ii) Analysis of temporal trends will help identify if the capacity of the environment to cope with dredging, and other anthropogenic pressures in the region is exceeded (see Barrio Froján et al., 2008[9]).
iii) Distinguish long-term trends (e.g. climate driven) from dredging impact?
iv) The data could also usefully contribute to UK monitoring programmes. Asking the aggregate industry to contribute, indirectly, to such initiatives has some logic given that their activities, in combination with other anthropogenic pressures, may have a bearing on the status of the UK seas.
v) The time-series would also provide a check on the health of surrounding faunal assemblages. This is important as these areas will have an important role, through provision of individuals and larvae, in the eventual recolonisation of impacted areas.
vi) With careful positioning such stations can also provide reassurance that dredging effects are not extending beyond the modelled SIZ.

3. Dedicated study sites

Within each region, a dedicated study site would be used to answer important questions concerning the effects of dredging (e.g. size of secondary impact zone, time required for physical and biological recovery). The results from this work will be used as a proxy for all similar extraction sites in a region, and once questions have been answered, the work will cease, with results published in the peer-review literature. Focusing effort and resources on a single site, rather than spreading effort across multiple sites (as is presently the case), is likely to provide a more robust understanding of dredging effects. The need to have a dedicated study site, and what question(s) need to be addressed will depend on what work has previously been undertaken in the region.

5.2 SURVEY STRATEGY

Like the existing approach, the RSMP requires sample collection for the purposes of characterisation, and then for monitoring. The survey work associated with each stage is described below.

5.2.1 Characterisation

Implementation of the RSMP approach requires a comprehensive regional baseline characterisation for fauna and sediments. Multiple sampling stations would be located within Primary Impact Zones (PIZ), Secondary Impact Zones (SIZ), Reference sites (REF) and Context areas (see Figure 7a). The Context areas are located outside the PIZ/SIZ, and Reference sites. Reference sites would be selected, using a variety of data sources (e.g. acoustic data, modelled biotopes) to represent all the faunal communities found with the footprint of dredging effect. One $0.1m^2$ Hamon grab sample would be collected from each station within the PIZ, SIZ and most of the REF and Context stations. At the remaining REF and Context stations, a small number of replicate samples would be collected. The data generated from these replicated sites will be used in subsequent monitoring. Once the faunal data have been analysed, a map of faunal distribution would be produced (Figure 7b).

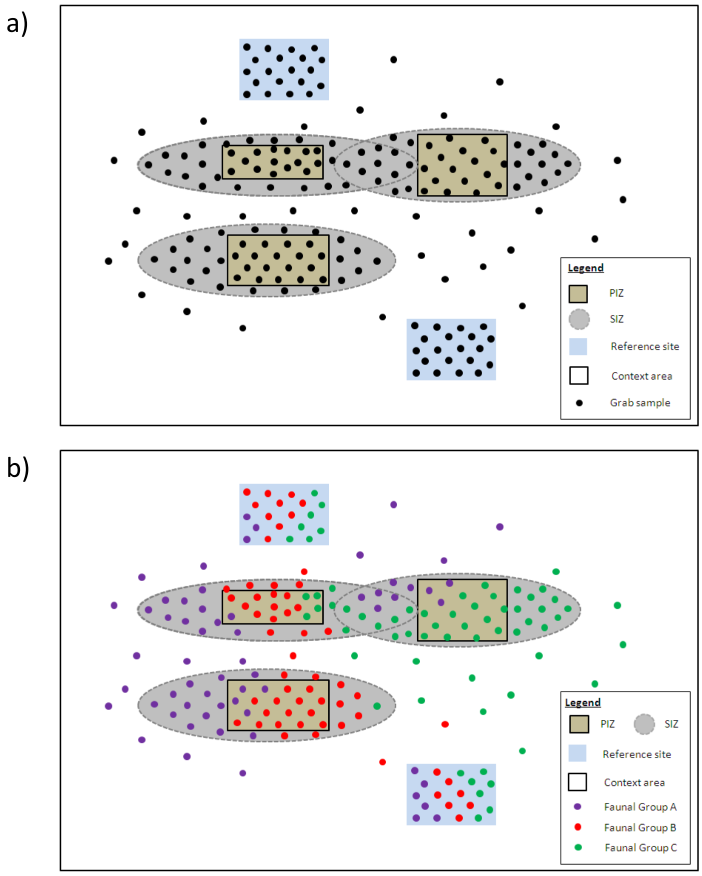


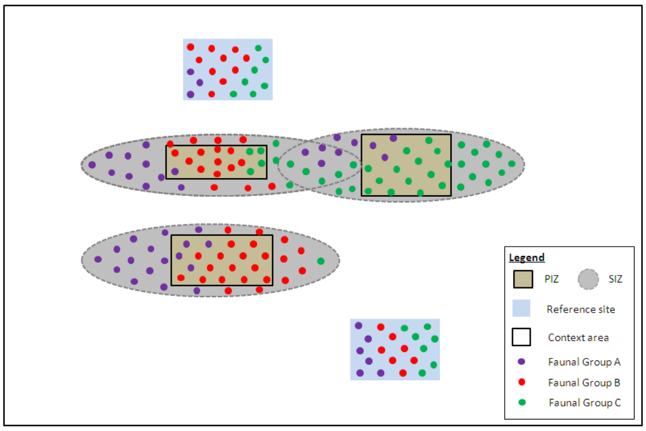
Fig. 7. Hypothetical survey design for characterisation of benthic macrofauna and sediments under the RSMP approach. Single samples are collected from each of the locations in part a. The faunal data from these sites are analysed to produce a map of faunal distribution (part b).

5.2.2 Monitoring

Monitoring would comprise two elements. The first is the monitoring of sediment composition within the PIZ, SIZ and REF sites (Figure 8a). The acceptability of changes in sediment composition within the PIZ and SIZ is determined according to the method in Cooper (2013)[1]. Essentially the sediment composition of monitoring samples is compared against an acceptable range for each sediment fraction (% coarse gravel, % medium gravel, % fine grave, % coarse sand, % medium sand, % fine sand, % silt/clay). The acceptable range will vary according to which of the faunal cluster groups was found at the stations during the baseline characterisation stage, and is simply the range of sediment composition naturally found in association with the group in the wider region. Where the proportion of any sediment fraction lies outside the acceptable range then this is termed a 'deviation'. The total number of 'deviations' within the PIZ or SIZ is then expressed as a percentage of the total number of possible deviations (i.e. assuming values of all sediment fractions were outside the acceptable limits). This value is subtracted from 100 to arrive at an overall value for percentage compliance. Changes in sediment composition within the PIZ or SIZ are deemed acceptable where the value of percentage compliance is equal to or greater than any of the values seen for the reference sites. Values of percentage compliance seen in the reference sites are assumed to be related to natural or non-dredging related change.

The second element is the monitoring of sediments and macrofauna at the network of long-term benthic monitoring stations (Figure 8b). In effect, the monitoring would be a repeat of the characterisation survey, except that samples acquired within the PIZ, SIZ and from most of the Reference site stations would not need to be processed for macrofauna. Unlike the present approach to monitoring, there would be no requirement for a separate pre-dredge survey.

a)



b)

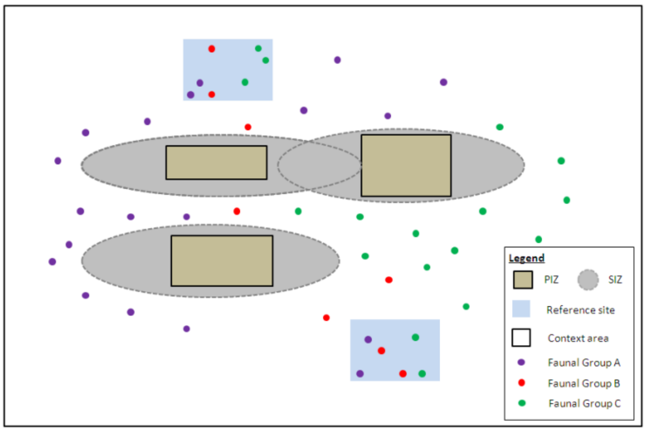


Fig. 8. The two elements of the monitoring programme under the RSMP. Part a shows the sample sites located within the PIZ, SIZ and Reference areas. These samples are used to monitor the acceptability of changes in sediment composition. Part b shows the long-term benthic monitoring stations. These sites are sampled for sediment and macrofauna.

5.3 UNDERPINNING SCIENCE

The rationale for each of the three elements of the RSMP approach (see Section 5.1) is based on scientific evidence. In this section this evidence is briefly summarised, with links made to other sections in the report where further information can be found.

1. Acceptable change limits for sediment particle size

In recent years, attempts have been made to try to understand the significance of localised impacts of aggregate dredging for the wider ecosystem. For example, Kenny et al. (2010) found similar trends over the period 1983 to 2007 in various aspects of the ecosystem for the East Coast and the wider North Sea. Based on these findings, they suggested that the main driving force behind environmental change in the East Coast region is not dredging related. Daskalov et al. (2011) used a spatial dynamic food web model to look at the effects of reductions in benthos, within dredged areas of the eastern English Channel, for fish. Whilst their results showed a localised decline in catches of demersal fish and shellfish, they predict that such effects are likely to be mainly localised to the vicinity of the extraction sites. Whilst these two studies do provide some reassurance, we should recognise that further work is required understand the wider significance of dredging in all dredging regions. Work is also required to validate the model predictions made by Daskalov et al. (2011). Finally, Pearce (2008) looked at the diet of demersal fish species in and around extraction sites from all major dredging regions. Her work suggests that most fish species are generalist feeders. Therefore, whilst the seabed within active extraction areas may become less attractive due to depletions of benthic food sources, fish are likely to be able to exploit alternative sources of food from outside the area.

Given the incomplete state of knowledge with regards to wider ecosystem effects of dredging, it is sensible to remain precautionary, and, in so far as is possible, to seek the return of the original faunal assemblage after dredging. Such an approach will help ensure the sustainability of marine aggregate dredging for seabed faunal communities and the wider ecosystem.

Work undertaken to understand recovery of the seabed following marine aggregate dredging suggests that faunal recovery is dependent on the physical condition of the seabed. Where the habitat remains largely unchanged then a full faunal recovery is a realistic expectation (Cooper et al., 2007b[10]). In contrast, where the habitat left by dredging is very different, then a faunal recovery to a pre-dredge state may be prolonged (Wan Hussin et al., 2012[4]) or even unrealistic (Desprez, 2000; Barrio Froján et al., 2011[6]). These results should not be taken to imply that any physical change is necessarily detrimental, as a number of studies have shown that some degree of physical change is not necessarily a barrier to full recovery (Hitchcock and Bell, 2004; Seiderer and Newell, 1999). Cooper et al. (2011b)[5] showed how faunal communities around the coast of England varied in their sensitivity to changes in sediment composition, and this suggests that we should be more and less concerned about physical changes resulting from dredging according of the identity of the pre-dredge faunal assemblage. In Cooper (2012)[3] and Cooper (2013)[1] it is shown how the limits of acceptable change in sediment composition can be identified by using broadscale regional datasets to identify the range of sediment composition naturally found in association with the pre-dredge faunal assemblage in the wider region. Clearly if physical conditions after dredging fall outside the acceptable limits of change then there needs to be some way of addressing the problem. This issue of restoration is addressed in Cooper et al. (2013)[2], which explores the range of techniques available. The high financial costs and practical difficulties associated with restoration (Cooper et al., 2013[2]; Cooper et al., 2011a[7]) should serve as a powerful incentive to avoid the need for it.

2. Long-term benthic monitoring stations

Long-term records of biological data are extremely valuable for documenting ecosystem changes, for differentiating natural changes from those caused by humans, and for generating and analyzing testable hypotheses (Wolfe et al., 1987). In the context of marine aggregate dredging, the utility of long-term time series data is highlighted in Barrio Froján et al. (2008)[9].

3. Dedicated study sites

Advances in our understanding of the impacts of marine aggregate dredging have typically come from research studies, where significant resources have been employed to confidently address particular questions (e.g. Newell et al., 2004). As we understand more about the variability of the environment (Eggleton et al., 2011), it becomes increasingly possible to use study sites as a proxy for other similar sites.

5.4 ADVANTAGES

5.4.1 Ability to differentiate between statistical and ecological significance of change

The ability to differentiate between statistical and ecological significance of changes in sediment composition has a number of benefits. Firstly, it allows the regulator to set an effective licence condition for acceptable change. This condition then provides a very clear focus for monitoring, namely to determine whether sediment conditions remain within acceptable limits or not.

Where conditions are outside the acceptable range then there is an opportunity for industry and the regulator to work together to identify a suitable management response to ensure conditions are brought back within acceptable limits. This opportunity arises each time monitoring is carried out. This process of adaptive management should help avoid the need for costly restoration efforts at the end of the licence term (Cooper et al., 2013[2]). However, were restoration ever required, then the acceptable change limits for sediment composition provide a clear target for restoration efforts, namely to bring sediment back within the acceptable range.

One of the benefits of the new approach is that it does not require sediments to be left in exactly the same condition post-dredging. This is important as the exploited resources are rarely, if ever, uniform with depth. This means that as the seabed is lowered different sediments may be exposed. Clearly the extent of the change will determine whether conditions will remain within acceptable limits. However, it may be possible to predict what changes in sediment composition will occur within a licence area using data from vibrocore samples. Vibrocores are routinely collected from within extraction sites by the developer to determine the quality of the sediment resources in 3 dimensions. The vibrocorer consists of a metal tube of approximately 6 x 0.2 m which is held within a supportive metal frame. When in position, the core is driven into the seabed by the corer's pneumatic vibrating head. Upon recovery, the core sample, held within an inner plastic liner, is removed from the core tube. Examination of the core sample may reveal differences in the nature of sediments with depth, and samples can be taken from different sediment bands for later analysis of particle size composition. This information can be used to help predict how surface sediments might change in composition as the seabed is lowered.

5.4.2 Survey design robust to changes in the location of dredging

The positioning of samples throughout the PIZ and SIZ means that survey designs are robust against changes in the location of dredging. The design also makes it possible to identify the precise location of unacceptable changes in sediment composition (Cooper, 2013[1]). Given a sufficient number of baseline samples, it may be sensible to only monitor those stations in the vicinity of the active dredge zone, and its zone of secondary effect.

5.4.3 Benefits of regional approach

Working with monitoring data at a regional level offers a variety of potential benefits. Firstly, it will allow for a regular snapshot of the condition of the seabed and associated faunal assemblages within the wider region. For this reason, the data and findings from monitoring are likely to be of interest to the UK government in connection with the Marine Strategy Framework Directive

(Directive 2008/56/EC). For example, maintaining the composition of seabed sediments within the acceptable change limits has relevance to the seabed integrity descriptor, whilst the monitoring of macrofauna within context and reference areas has relevance for the biodiversity descriptor. In relation to the macrofauna, we may increasingly be able to differentiate between natural and anthropogenic change as the time series develops (Barrio Froján et al., 2008[9]). Secondly, the data might also allow for the assessment of cumulative and in-combination effects, something which has hitherto not been possible. The need to undertake such assessments will become more important as the level of anthropogenic pressure increases around our seas.

For industry, a regional monitoring approach is likely to be much more efficient, with obvious benefits for time and money. However, it might also benefit the regulator in that monitoring data from all sites can be reported together. This will speed up the consultation and assessment process. It should also have benefits for consistency in terms of how the regulator judges the acceptability of change at individual extraction sites. In addition, a regional perspective will also helpful when it comes to assessing the significance of site specific impacts.

5.4.4 Dedicated study sites will provide robust answers

One of the criticisms of the existing monitoring approach is that it does not generate robust answers to the questions posed. This is due to the tension between what is required to get robust answers, and what is affordable from the industry perspective. A good example of this is to compare the sampling effort employed as part of an aggregates research project with that typically associated with routine monitoring. Often the questions are the similar, but the level of effort is hugely different. For example, Newell et al. (2004a) took 208 0.1m^2 Hamon grab samples at each of the two sites in their study, while a typical monitoring survey might only acquire in the region of 30 to 40 grab samples. As a result, monitoring often fails to deliver robust answers, leading to more monitoring.

Under the RSMP approach, the outstanding research questions would be properly addressed at a representative site within each region. Once the question(s) have been answered then work can stop. This approach has worked well in the Eastern English Channel, where the outputs of the research have provided the regulator with a much better understanding of the nature of impacts locally. Improved understanding of dredging impacts will obviously allow for better management decisions.

5.4.5 Reduced costs

The costs of environmental monitoring under the RSMP approach are expected to be much lower than those under the current approach. Reasons for this are: (i) All the monitoring can be undertaken by one vessel, with obvious savings in terms of vessel mobilisation/demobilisation; (ii) It will only be necessary to process samples within the PIZ/SIZ for macrofauna at the baseline characterisation stage. After this, the stations within these areas will only need to be sampled for sediment particle size composition; (iii) Unlike the present approach, it will not be necessary to undertake a separate pre-dredge survey as the survey design for characterisation is the same as monitoring; (iv) The very specific question posed by monitoring means that the monitoring reports will be quicker to produce, and review; (v) the overall number of reference samples is lower as the data is effectively shared between all extraction sites. In time, it might also be possible for reference sites to be shared between different industrial sectors operating within the same region.

5.5 OUTSTANDING ISSUES

Whilst there is a momentum towards implementation of the RSMP approach, some important issues remain to be addressed. In this section I outline each of these issues, and, where possible, make suggestions for how they should be dealt with.

5.5.1 Challenge of establishing a baseline characterisation

Task

Implementation of the RSMP approach will require the production of a regional baseline characterisation for macrofauna and sediments. In the Eastern English Channel this characterisation was based on the results of a one-off regional survey undertaken before dredging started in 2007 (see Cooper, 2013[1]). Unfortunately, similar pre-dredge baseline characterisations do not exist for other dredging regions in the UK. This is because aggregate dredging has been ongoing in these regions for far longer, and, for many of the older extraction areas which were licensed under a different regulatory framework, there was never a requirement for the collection of baseline data. As a result, it will be necessary to construct a ‘baseline’ characterisation for these regions using a combination of new and existing data. Use of existing data will inform about an earlier condition of the seabed. Without this perspective we could run the risk of accepting a gradual decline in environmental condition as a result of a shifting baseline. In addition, the use of existing data will keep the costs of producing the baseline characterisation to a minimum.

Sources of existing data

Existing macrofaunal and sediment particle size data are available from a variety of sources. The most important of these will be the Regional Environmental Characterisation (REC) (James et al., 2010; Emu Ltd., 2009; Limpenny et al., 2011; Tappin et al., 2011) and Marine Aggregate Regional Environmental Assessment (MAREA) datasets (MAREA, no date) which have been produced for the Humber, East Coast, Thames, and South Coast dredging regions. In addition, individual developers hold similar data from the characterisation and monitoring surveys undertaken at contemporary extraction areas. Other sources of data include: (i) The Crown Estate's Marine Data Exchange (http://www.marinedataexchange.co.uk/), which allows access to the extensive benthic datasets produced by offshore windfarm developers; (ii) Cefas's benthic data holdings which include datasets collected under various research projects; and (iii) The Data Archive for Marine Species and Habitats (DASHH) (http://www.dassh.ac.uk/aims.html), which allows access to benthic datasets archived by different organisations.

New sampling stations

New macrofaunal and sediment particle size samples will be required for: (i) the new shared reference areas, (ii) ensuring adequate spatial coverage within the context areas (i.e. parts of the region outside the footprint of dredging effect (PIZ/SIZ)), and (iii) filling in any gaps in the spatial coverage within the PIZ and SIZ. Identification of new sampling sites within the PIZ/SIZ will be achieved by overlaying the respective polygons, in a geographic information system (GIS), on the existing sample stations layer. Where necessary, new sampling stations will be chosen to ensure good spatial coverage, with a minimum of 20 stations within each zone. To make this process easier, a series of sampling grids have been produced. The grids are all based on a triangular matrix, which is considered to be optimal for detection of patches (Barry & Nicholson, 1993). There are 8 grids in total. Four of these grids are intended for use within the PIZ, with the remaining four grids for use within the SIZ. The four PIZ grids are at a higher density than those used in the SIZ, reflecting the greater risk of sediment change within this zone. Different grids are provided for use due to the wide variety of sizes and shapes of PIZs and SIZs. No one grid would be suitable for use in all circumstances, and it is necessary to identify the most appropriate grids to provide at least 20 stations within each zone (PIZ/SIZ). Figure 9 shows a hypothetical survey design for a site where existing sample data are available. In this case, existing sample stations simply replace some of the stations in the regular grid.

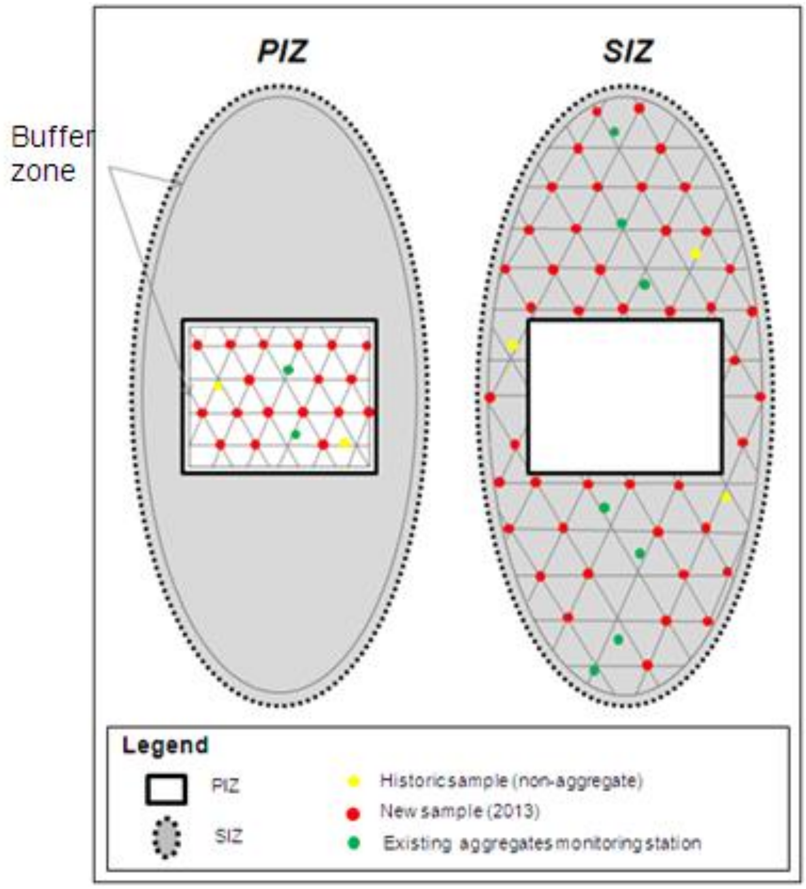


Fig 9. Proposed survey design for collection of samples within the PIZ/SIZ of licence areas.

Where existing data from within a PIZ or SIZ are to be used for the purposes of characterisation, it is recommended that a contemporary sediment particle size sample is obtained. These samples would be retained for 'insurance' purposes and would only be processed where unacceptable changes in sediment composition had been reported from the initial monitoring campaign. This additional sample would make it possible to check whether or not the change in sediment composition had occurred since the onset of dredging under the most recent licence.

Integration of new and existing data

Integration of new and existing macrofauna and sediment particle size data is likely to present a variety of challenges due to differences in data quality, sample collection and processing methodologies, and natural temporal changes. In deciding how to address these important issues it will be sensible to consider what it is we are trying to achieve. Fundamentally we will be seeking to present a macrofaunal assemblage characterisation which reflects the persistent differences found across the region, and not the stochastic variations associated with natural cyclical changes. It is likely that a variety of different approaches may need to be employed to achieve this aim. These may include, *inter alia,* the truncation of data (i.e. reviewing the species list to address inconsistencies in faunal identification between surveys), removal of rarer species, and analysis of data at higher taxonomic levels (e.g. genus or family).

Identification of final faunal groups

In seeking to identify broadscale faunal assemblages, one possibility would be to make use of the European Nature Information System (EUNIS) habitat classification system (European Environment Agency, 2004), or the JNCC's marine habitat classification system (Connor et al., 2004). The problem with these top down classification approaches is they make assumptions that faunal communities are aligned with particular substrate units, ignoring evidence to the contrary (Zajac et al., 2000; Hewitt et al., 2004; Stevens and Connolly, 2004; Zajac, 2008). For this reason it is intended to use a bottom-up approach (i.e. beginning with the raw faunal data) so that no assumptions are made about the relationship between fauna and sediment composition. However, because acceptable change limits are strongly influenced by the choice of faunal assemblage groups, it is important to consider how groups will be identified.

Faunal groups are typically identified using a clustering approach, although ordination can also be helpful, particularly where there is a steady gradation in community composition across sites (Clarke & Warwick, 2001). It is, however, important to recognise that clustering and ordination are simply exploratory techniques and that the identification of groups involves a degree of subjectivity. For this reason, it is important that individuals engaged in this task have some knowledge of the data and the specific issue being addressed. A more objective means of identifying faunal assemblages from clustering include the SIMPROF routine (Clarke et al., 2008). This tool is found in the Primer-e package (Clarke & Warwick, 2001) and tests for the presence of sample groups (or more continuous sample patterns) in *a priori* unstructured sets of samples. Whilst useful, the SIMPROF routine can yield unwieldy numbers of cluster groups with large datasets, and there is still a need to judge whether the groups identified are ecologically meaningful. It is for this reason that a different clustering approach was used in Cooper (2013)[1]. In this study clustering was performed in R (R development Core Team, 2010) using the k-means R function available from the flexclust library. The Hartigan and Wong (1979) algorithum was used to find solutions based on different numbers of pre-defined cluster groups. Implications of choosing different numbers of cluster were explored by plotting the number of cluster groups against the minimum within cluster sum of squares (Everitt, 2005) or a pseudo F-statistic (Calinski and Harabasz, 1974). In the example shown in the Figure 10, the plots suggest that the reduction slows down when there are around 8 to 10 clusters. Clearly it will be important that decisions taken with regards the identification of faunal groups are transparent and defensible, and, where possible, decisions should involve a range of stakeholders. While the assemblages identified will be used to produce acceptable limits of change for sediment particle size, the map will also be extremely useful for the planning and management of all offshore activities which affect the seabed. For this reason there is an argument for producing an over-arching national classification of assemblages, within which all assessments can be made, rather than a series of ad hoc and incompatible analyses on a case by case basis.

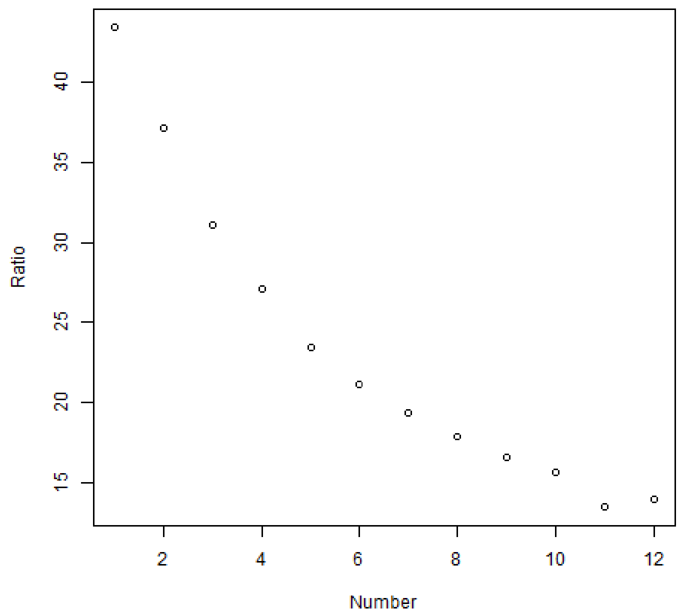


Fig. 10. Pseudo F-Statistic for each number of clusters.

5.5.2 Dealing with new areas not included in the baseline characterisation

Whilst the baseline characterisation exercise will allow the faunal groups present within existing licence areas to be identified, there is an issue of how to integrate future new sites into the characterisation. One possibility is to use a statistical procedure to match up new benthic data with the original baseline faunal groups. Where the faunal composition of new samples appears to be different from all the original baseline groups then it may be possible, in theory, to re-cluster using the combined dataset, with acceptable change limits in sediment composition determined from this new assessment.

5.5.3 Approach does not deal with sensitivity of individual species

Some concern has been expressed about the extent to which the new approach can assure the sediment composition requirements of individual species. In my view it is unrealistic to try to manage the seabed for all macrofaunal species. We should of course have regard for the habitat requirements of individual species, but these considerations should be made at the characterisation stage. If necessary, the boundaries of the extraction area can be modified to exclude particularly sensitive species or features, or the project can even be rejected altogether. Once a decision has been made to allow dredging to go ahead then the focus of management should be about maintaining the integrity of the environment in a broad sense (i.e. at a community level). However, the faunal characterisation dataset could be used to improve our understanding of the distribution of individual species, and hence their vulnerability to dredging.

5.5.4 Influence of other variables

Dredging can be responsible for changes in sediment composition and bathymetry. Changes in bathymetry can result in changes to hydrodynamics, and hence this factor may also influence what faunal communities are likely to recolonise an area after dredging. For this reason, it is sensible to try to integrate other depth related variables into the approach to improve our ability to predict what faunal community which is likely to recolonise post dredging. There is an intention to do this using a statistical modelling approach based on Maximum Likelihood Classification (Cooper and Barry, in-prep). Using this technique, half the available data are use to train the model, with the remaining data then used to test the model's ability to predict which faunal group will be present based on the inputted environmental variables. A further benefit associated with this modelling approach is that it will reduce the need for expert judgement in terms of identifying what faunal community is likely to return after dredging. This will be particularly useful in cases where sediment composition has gone beyond the limits of acceptable change, but where it is not obvious what the changes will mean for faunal recovery.

5.5.5 Sample outliers

In the Eastern English Channel it remains a hypothesis that a full faunal recovery will occur where the composition of sediments is left within the limits of acceptable change (Cooper, 2013[1]). For this reason, the aggregates industry will be required to initiate a study of faunal recovery at a dedicated study site (see Section 5.1). Were a full faunal recovery not to occur, despite sediments remaining within the acceptable range, this would suggest that the limits of acceptable change would need to be modified. One reason this situation might arise is if the initial upper and lower limits of acceptable change were unduly influenced by outliers. In anticipation of this issue, Cooper (2012)[3] used a 95th percentile range to define the limits of acceptable change in sediment composition. The same approach was initially taken during the EEC study (Cooper, 2013[1]). However, it was later rejected due to the presence of an 'outliers' in PIZ of one site. In this situation sediments breech the acceptable limits even before dredging starts. Clearly this is nonsensical, so the approach was rejected in favour of using the whole sediment envelope (Cooper, 2013[1]). There are two competing issues here. On the one hand you want to ensure that you capture the full range of sediment composition which will support the faunal assemblage. On the other hand you do not want the outliers to wrongly overestimate the tolerance limits. One solution to this might be to use the 95th percentile approach, but to extend the tolerance limit for a particular site if an outlier was present in that location.

5.5.6 Frequency of Monitoring

It is important to draw a distinction here between the monitoring undertaken to assess changes in sediment composition, and the monitoring undertaken at the long-term benthic monitoring stations. For the long-term benthic monitoring stations, surveys should be carried out on a regular basis, possibly every two years. For the sediment monitoring, a sensible approach might be to have more frequent monitoring until it is established that unacceptable changes in sediment composition are not developing. However, as the risk of unacceptable changes occurring at older sites where licences have been renewed is, by definition, much lower, it may be sensible to accept a lower frequency of monitoring at these sites. In a sense, the risk is with the developer, as they are required to comply with the licence condition at the end of the licence term. More frequent monitoring will provide an early indication of a problem, and this will allow for more time to address it.

5.5.7 Development of nature conservation features following the baseline survey?

Under the existing approach to monitoring (Ware & Kenny, 2011), the benthic macrofauna are periodically monitored during the term of the extraction licence. Where monitoring data shows that species or features of nature conservation are present within the footprint of effect (PIZ/SIZ) then there is an opportunity for appropriate mitigation to be put in place. For example, exclusion zones have, in the past, been established around areas of *Sabellaria spinulosa* reef. Where present, the reef formed by this species of polychaete is afforded protection under a raft of legislation including the Habitats Directive (Council Directive 92/43/EEC) and the Natural Environment and Rural Communities Act (NERC) 2006.

As macrofauna will not be monitored within the PIZ/SIZ under the new approach (Cooper, 2013[1]), concern has been expressed that reef features might not be afforded the same degree of protection. I take the view that the appropriate time to establish exclusion zones is at the EIA stage, with decisions informed using the characterisation dataset. Of course, reef features can develop within the footprint of dredging effect once the site has been licensed. However, in these cases the reef could be considered to be: (i) ephemeral (Limpenny et al., 2010), (ii) tolerant to the impact (Last et al., 2011), or (iii) possibly even present as a result of the impact itself; a result of the increased sand supply (Pearce et al., 2007). Clearly, reef present within the PIZ might well be vulnerable to dredging, but given the above, I suggest that the setting up of exclusion zones in response to monitoring is perhaps over-precautionary.

5.5.8 Sediment stratification

One drawback of the particle size data obtained from a Hamon grab is that it does not necessarily tell us how sediments are arranged on the seabed (Cooper et al., 2011a[7]). For example, your particle size data can suggest the seabed is characterised by a mixed sandy gravel when if fact you have sand overlying gravel, or gravel overlying sand. Whilst this does not normally occur, it highlights the need for other datasets to 'sense check' the sediment data. This could be done using acoustic and/or underwater images.

5.5.9 Sediment heterogeneity

Another concern relates to heterogeneity, or small scale patchiness, in sediments and benthos. It is theoretically possible that the sediment composition of a monitoring sample may fall outside the acceptable change limits as a result of small-scale patchiness in the sampled seabed. This issue may be addressed by reference sites, on the assumption that they are equally as heterogeneous. However, I do not think this is likely to be a major issue as the clustering of benthic data produces a relatively broadscale coarse characterisation of faunal assemblage distribution. Another option would be to collect one or more replicate samples from each sampling station at the initial characterisation stage. In order to minimise costs, the replicate sample(s) would only be processed for sediment particle size. If the sediment particle size composition of the replicate samples was outside the limits for the initial sample (processed for macrofauna and sediment particle size) then this would indicate that breeches identified during subsequent monitoring should be treated with caution (i.e. they could be a result of the small-scale heterogeneity in sediment composition as apposed to dredging).

5.5.10 Perceived positive change

Under the RSMP approach, any change in faunal group is flagged as being a potential problem. However, there are some circumstances where the changes in sediment composition could be regarded as potentially positive. For example, in the case of 'reverse screening', where coarser sediment fractions are rejected in order to obtain a sand cargo. In theory, this practice could increase the proportion of gravel on the seabed, possibly leading to the eventual recolonisation by a more diverse faunal community. For this reason it will be important to consider the direction of any change when judging acceptability. That is not to say such increases in diversity should always be welcomed. This has to be a judgement call, and knowledge of the regional extent of lost habitat will inform such decisions.

5.5.11 Representativeness of sediment particle size samples?

In fine sediments, it is possible to get a reasonably accurate estimate of sediment particle size distribution using a small volume of material. However, as the size of particles increases, so does the volume required to obtain a representative particle size distribution (British Standards Institution, 1996; Passchier, 2007). Despite these facts, the size of sediment particle size samples is standardized to approximately 0.5 litre (Ware and Kenny, 2011). The reason for this is that sediment samples are taken from the faunal grab as a sub-sample, and if the volume of the sediment sub-samples were to differ, or be any larger, then it would compromise the comparability of the faunal data. In theory it would be possible to take the sediment particle size sample from a separate grab deployment, and this has been advocated for some monitoring programmes (Mason, 2011). However, there are a number of potential problems with this approach. Firstly, you lose the direct comparability with the faunal data. Secondly, it requires someone to make a subjective judgment as to the likely representativeness of the sediment grab. Thirdly, it can be difficult for a boat without a dynamic positioning system to maintain its position when making multiple grab deployments. Also, as we are making use of historic data, we have to maintain the comparability of datasets where sediments have been acquired as a sub-sample from the faunal grab. However, in relation to grab samples where faunal data is not required (e.g. stations sampled within the PIZ and SIZ for the purposes of monitoring the acceptability of changes in sediment composition), an option exists for either taking a larger sub-sample, or even using the entire sample for sediment particle size analysis.

5.5.12 Are the pre-defined acceptable change limits correct?

Where the composition of sediments after dredging remains within the limits of acceptable change (see Cooper, 2013[1]), then the assumption is that a full faunal recovery will occur. This hypothesis will require testing, however, and it is recommended that the aggregates industry should initiate appropriate studies as soon as suitable opportunities arise. This question of recovery will be a priority issue to be addressed at the dedicated research sites (see Section 5.1). To properly test the hypothesis that faunal recovery will occur if sediments remain within the acceptable limits of change it will be necessary to monitor recovery in response to a range of different sediment conditions, both where these are within and outside the acceptable range. Clearly it may take a considerable amount of time to gain this understanding, given that recovery would typically be expected to take a number of years. One way to get answers more quickly would be to employ an experimental approach, possibly using defaunated sediment trays to assess faunal in the locality (e.g. Collie et al., 2009; Guerra-Garcia and Garcia-Gomez, 2006).

5.5.13 Statistical Power

Where data are available, statistical power analysis will be used to determine the required number of samples to detect a specified level of change in a particular parameter. For the long-term benthic monitoring stations (see section 5.1), power analysis will be used to determine the number of samples required to detect a specific change (possibly 10%) in the number of species over a 5 year period. This assessment will be made for each of the faunal assemblages identified in the baseline characterisation. The results will then be used to select a subset of monitoring stations from each faunal group within the context and reference areas.

5.5.14 Consideration of functional differences in faunal communities

Under the approach described in Cooper (2013)[1], changes in sediment conditions which could lead to a change in macrofaunal community are considered undesirable. In these cases expert judgement would be required to determine the acceptability of the change. To help in this judgement it is proposed to use the faunal characterisation data to construct a similar characterisation map based on trait expression, as a proxy for ecosystem function (see Cooper et al., 2008[8]). This will provide the regulator with a better insight into the significance of any changes in faunal group identity at individual sampling stations. For instance, there is likely to be less concern about changes in faunal group where the new group appears to be functionally similar to the original.

6. CONCLUSIONS

The purpose of this thesis was to critically examine the case for switching to a new method of monitoring the environmental impacts of marine aggregate dredging on the seabed. This was done by examining both the existing monitoring approach (Ware & Kenny, 2011), and the new Regional Seabed Monitoring Plan (RSMP) approach (Cooper, 2012[3], 2013[1]). The essay also considered how the findings from past research, particularly the studies I have personally been involved with, have contributed to the development of the new monitoring approach. The different facets of the thesis are set out in a conceptual model (Figure 10). The model also highlights the benefits which are expected to result from switching to the RSMP approach.

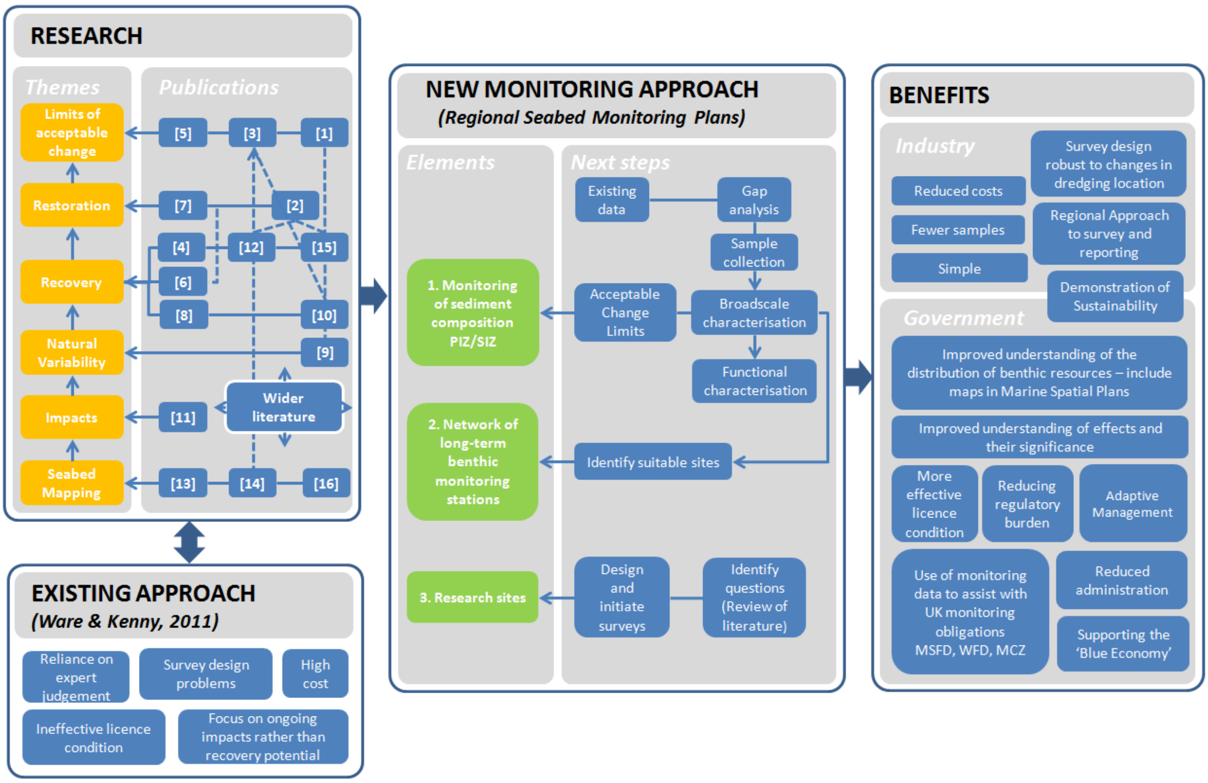


Fig. 11. Conceptual model showing how the findings from research, combined with experience of the existing monitoring approach, have led to the development of the new Regional Seabed Monitoring Plan (RSMP) approach. The model also highlights the benefits which are expected to arise from the RSMP approach.

Research themes have followed a logical progression of ideas. Initially work focused on the impacts of dredging (Cooper et al., 2007a[11]). This led to questions concerning recovery (Boyd et al., 2003[15], 2005[12]; Cooper et al., 2007b[10], 2008[8]; Barrio Froján et al., 2011[6]; Wan Hussin et al., 2012[4]). When it became apparent that faunal recovery may, in some cases, be prolonged or even unrealistic

(Barrio Froján et al., 2011[6]; Wan Hussin et al., 2012[4]), attention turned to active seabed restoration, to determine whether there were actions that could be taken to try to promote physical and biological recovery of the seabed. When results of a gravel seeding experiment (Cooper et al., 2011a[7]) showed some success, the issue of when it might be appropriate to intervene became relevant. Due to the difficulties in undertaking a cost-benefit assessment for restoration (Cooper et al., 2013[2]), the need for setting criteria for acceptable change in the physical condition of the seabed became apparent. Eventually, work on the sensitivity of faunal communities to changes in sediment composition (Cooper et al., 2011b[5]) and habitat mapping (Brown et al., 2002[16], 2004a[13], 2004b[14]) led to the development of acceptable change limits in sediment composition (Cooper, 2012[3], 2013[1]). A combination of the acceptable change limits approach, and work on natural variability (Barrio Froján et al., 2008[9]) led to the concept of the RSMP approach.

The most obvious difference between the existing and RSMP approaches is the different questions they seek to address. The existing monitoring approach seeks to answer the question *'What is the impact of ongoing dredging on sediments and fauna?',* and to confirm that the observed impacts are no greater than what was predicted in the Environmental Statement. I argue this is the wrong question because: (i) these impacts are now reasonably well understood, and (ii) the monitoring results do not tell us what is likely to happen to the site after dredging (i.e. faunal recoverability). At present, judgements concerning recovery potential are typically a matter of subjective 'expert judgement'. If we are to ensure the long-term sustainability of dredging for seabed macrofaunal communities, then the question for monitoring must be *'Will the affected seabed be able to support the return of the original faunal assemblage'*; this question is central to the new monitoring approach described in Cooper (2013)[1]. This new approach is consistent with the principles of sustainability, and recognises that we do not yet fully understand what the implications of changes in seabed macrofauna are for the wider ecosystem, particularly in the context of cumulative effects.

The new approach (Cooper, 2012[3], 2013[1]) works by identifying the range of sediment particle size composition naturally found in association with the pre-dredge faunal assemblage in the wider region. Theoretically, so long as sediment composition remains within this range then it should be possible for the return of the original faunal assemblage after dredging. It is this new understanding of the relationship between sediments and faunal composition that allows us to pursue a new approach to monitoring using sediments as a proxy for likely faunal recovery. The ability to differentiate between statistically significant changes in sediment composition and the likely future ecological significance allows us to set meaningful licence conditions. It also provides a clear focus for monitoring. Where monitoring shows sediment conditions fall outside the acceptable range then there is an opportunity for management intervention to bring conditions back within acceptable limits (i.e. adaptive management).

The first step in implementing the RSMP approach is to construct a regional 'baseline' characterisation for macrofauna and sediments. With the exception of the Eastern English Channel, where we already have such a characterisation, this will involve the integration of existing and newly acquired data. New samples will be required within new regional reference sites, and to ensure adequate spatial coverage within context areas and the PIZ/SIZ of individual extraction sites. Integration of data collected at different times, and processed by different laboratories is expected to present some challenges, but the aim will be for the resulting map to reflect the persistent differences in faunal communities across the region, rather than stochastic differences associated with natural variability or the identity of the processing laboratory. As the limits of acceptable change in sediment composition are so dependant on the faunal groups identified from the characterisation it will be important that decisions taken in identifying these groups are transparent, and, where possible, involve input from other relevant stakeholders. In the longer term, it is recommended that the faunal data should be used to construct a characterisation based on trait expression, as a proxy for ecosystem function. This will provide the regulator with a better insight into the significance of any changes in faunal group after dredging. From the baseline dataset, a suitable number (to be determined using power analysis) of stations will be selected as long-term benthic monitoring stations. Finally, a review of the literature will be undertaken to identify what questions need to be addressed at the dedicated research sites. It is likely that questions concerning recovery will be particular priorities for these sites, given that it remains a hypothesis that faunal recovery will occur where sediment composition remains within the acceptable limits of change. Finally a statistical modelling approach will be developed to improve predictions of what faunal communities will recolonise after dredging, based on a variety of different physical variables.

Adoption of the RSMP approach is expected to offer a range of benefits for both the aggregates industry and the regulator. For the industry, the approach will reduce the complexity and costs of monitoring. Cost savings will result from the collection of fewer macrobenthic samples, and also from the regional approach taken to sampling. For the regulator, the use of acceptable change limits, combined with a regional approach to monitoring, will allow for much more effective environmental protection. In addition, the data generated by industry, whilst serving their own specific requirements, could also be useful to help fulfil other government objectives concerning monitoring (e.g. demonstration of Good Environmental Status (GES) under the seabed integrity and biodiversity descriptors of the Marine Strategy Framework Directive (European Commission, 2008)).

Both parties will benefit from the new benthic survey design which caters for changes in the location of dredging within the licensed area. Finally, the very clear purpose of benthic monitoring, combined with the regional approach will allow for a much more streamlined reporting and review process. As a result of these expected benefits, the RSMP approach has attracted universal support from all stakeholders.

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Paper #1

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Setting limits for acceptable change in sediment particle size composition: Testing a new approach to managing marine aggregate dredging

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ABSTRACT

A baseline dataset from 2005 was used to identify the spatial distribution of macrofaunal assemblages across the eastern English Channel. The range of sediment composition found in association with each assemblage was used to define limits for acceptable change at ten licensed marine aggregate extraction areas. Sediment data acquired in 2010, 4 years after the onset of dredging, were used to assess whether conditions remained within the acceptable limits. Despite the observed changes in sediment composition, the composition of sediments in and around nine extraction areas remained within pre-defined acceptable limits. At the tenth site, some of the observed changes within the licence area were judged to have gone beyond the acceptable limits. Implications of the changes are discussed, and appropriate management measures identified. The approach taken in this study offers a simple, objective and cost-effective method for assessing the significance of change, and could simplify the existing monitoring regime.

1. Introduction

The UK marine aggregate dredging industry provides sand and gravel to domestic and European customers for construction and coastal defence (Highley et al., 2007). Material is extracted from the seabed using purpose-built dredging vessels, with operations taking place within 70 licensed areas located around the coast of England and Wales (Russell, 2011). In some locations, aggregate dredging has been shown to alter the composition of seabed sediments (e.g. Dickson and Lee, 1972; Kenny and Rees, 1996; Kenny et al., 1998; Newell et al., 1998, 2004a; Boyd et al., 2002; Cooper et al., 2007). Such changes can occur in a variety of ways (see Newell et al., 1998), although a major cause is associated with sediment screening (Poiner and Kennedy, 1984; Hitchcock and Drucker, 1996; Newell et al., 1998, 2004a), a process used to modify the composition of dredged cargoes, resulting in the return of unwanted sediment fractions, normally sands, to the seabed.

Research suggests that changes in the composition of seabed sediments can affect the ability of a site to recover, in terms of the benthic fauna, to a pre-dredge state post-dredging (Desprez, 2000; Newell et al., 2004a,b; Boyd et al., 2005; Robinson et al., 2005; Desprez et al., 2010; Cooper et al., 2011b,c; Barrio Froján et al., 2011; Wan Hussin et al., 2012). The composition of seabed sediments is also important for other components of the ecosystem including herring spawning success (de Groot, 1980). To mitigate the effects of dredging, conditions are often applied to extraction licences. Examples of licence conditions include: (1) limits on the extraction rate; (2) limits on the total tonnage extracted; (3) restrictions regarding the quantity of material which can be screened; (4) a requirement to leave the seabed in a similar physical condition after dredging; and (5) a requirement to monitor the environmental effects of dredging over the licence term (see Ware and Kenny, 2011).

The challenge for both the developer and the regulators is identifying, from the monitoring programme, what constitutes unacceptable environmental change. The reason this can be difficult is that monitoring looks at changes in response to ongoing dredging with, typically, little or no information about how long effects will last (i.e. recoverability). Despite some efforts (see Foden et al., 2009; MESL, 2007), knowledge of recovery times is still partial. In addition, our understanding of the wider significance of localised environmental change is not well understood (e.g. Kenny et al., 2010; Daskalov et al., 2011). For these reasons, decisions regarding acceptability of change are typically based on expert judgement. Whilst the licence condition requiring sediments to be left in 'similar' physical condition (ODPM, 2002) is sensible, given the implications for faunal recovery, the subjective nature of the term 'similar' means that the condition is of little practical use (Cooper et al., 2011a). If government policy makers, regulators

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and industry are to achieve their shared goal of sustainability (BMAPA, 2006; UK Marine and Coastal Access Act, 2009) there needs to be a better way of differentiating between acceptable and unacceptable environmental change.

A possible solution to this problem was recently proposed in Cooper (2012). His approach works by identifying the range of sediment particle size composition naturally found in association with the pre-dredge faunal assemblage(s) in the wider region. Theoretically, as long as sediment composition within areas of impact remains within this range, which can be specified as a licence condition, then it should be possible for a return of the pre-dredge faunal assemblage after cessation of dredging. This approach offers a number of advantages:

1. It has a clear scientific rationale, with the aim of maximising the sustainability of marine aggregate dredging.
2. The local environment is used to define the limits of acceptable change. This is important given results in Cooper et al. (2011b) which showed that benthic faunal communities are not uniformly sensitive to changes in sediment composition, with lower sensitivity in high energy sandy areas, and higher sensitivity in low energy, gravel areas.
3. It allows for change in sediment composition as a result of dredging. This is important given that some degree of change is highly likely given that targeted resource deposits are rarely, if ever, uniform in composition.
4. As changes in sediment composition are easily measurable, this means that it should be clear when conditions are not within acceptable limits, allowing for an appropriate management response (see Cooper, 2012).
5. It has the potential to reduce the costs of monitoring programmes by focusing on sediments rather than macrofauna.

With the above approach, there is still a need to understand the capacity for physical and biological recovery. In addition, there will continue to be a need to monitor the macrofauna at context stations (within areas outside the predicted effects of dredging). These areas are likely to have an important role in the recolonisation of dredged areas upon cessation of dredging, and for allowing the regulator to assess whether the level of anthropogenic pressure in the region is sustainable (see Barrio Froján et al., 2008).

A trial of this new approach to the setting of acceptable limits of change in sediment composition was undertaken using data from an extraction site off Hastings on the south coast of the UK. This study (Cooper, 2012) showed that sediments within the licence area remained within a pre-defined acceptable range. The expected faunal recovery potential of the site was confirmed by results in Cooper et al. (2007), who reported a 7 year recovery time within areas of low dredging intensity. Given the advantages of the approach, it was concluded that it should be considered for use in the regulatory context. However, before this could happen, there was an obvious need for further testing and refinement of the method.

The aim of the present study was to test the approach in the eastern English Channel (EEC), a region containing ten aggregate extraction areas. The EEC was chosen due to the availability of extensive baseline and monitoring datasets, and a desire on the part of the developers to review the existing monitoring regime (ECA, 2011). Specific objectives were to: (1) Identify, characterise and map the broadscale distribution of macrofaunal assemblages present in the survey area; (2) Identify the range of sediment particle size composition found in association with each assemblage; (3) Identify the macrofaunal assemblage(s) present within each of the extraction sites, and their associated zone of potential secondary effects; (4) Identify a suitable licence condition for acceptable change in sediment composition for each licensed area; and (5) Assess compliance with the stated condition using the most recently available monitoring data from 2010, 4 years after the start of dredging operations.

2. Methods

2.1. Data

The baseline dataset used in this study came from the 2005 Eastern English Channel Regional Environmental Assessment (REA) survey (ECA and EMU Ltd., 2010a, 2010b). This survey included 458 samples for macrofauna and sediments. Macrofaunal samples were processed over a 1 mm sieve, and the resulting data included countable, and non-countable colonial taxa. The sediment particle size data were supplied as percentage weight by size class (<0.063 mm, 0.63 mm, 0.125 mm, 0.25 mm, 0.5 mm, 1.0 mm, 2.0 mm, 4.0 mm, 8.0 mm, 16.0 mm, 32.0 mm, ⩾64.0 mm). Whilst other baseline benthic datasets from the region were available (e.g. James et al., 2007; ECA and Emu Ltd., 2010c, 2010d), issues of comparability precluded their use. Monitoring data from 2010 (ECA and EMU Ltd., 2010e) included 427 sediment samples, and these data were used to assess for change in sediment composition after 4 years of dredging. Samples from both surveys were acquired using a 0.1 m^2 Hamon grab, and were processed in a comparable way (see Ware and Kenny, 2011). The location of baseline and monitoring stations is shown in Fig. 1.

2.1.1. Treatment categories

All samples were assigned to one of the following treatment groups, depending on their location:

Primary Impact Zone (PIZ). Samples taken from within the licence boundary, and which may or may not have been subject to the direct effects of dredging.

Secondary Impact Zone (SIZ). Samples taken outside the PIZ, but within a full tidal excursion of the licence boundary. The SIZ is subdivided into *near-field* (within 2.5 km of the licence boundary), and *far-field* (>2.5 km to the full tidal excursion) zones. Samples intersecting more than one SIZ were also assigned to a 'cumulative' category.

Reference. Samples taken from stations located beyond the predicted effects of dredging (i.e. outside the PIZ and SIZ). This category includes samples taken from within defined references boxes, or positioned throughout the remainder of the survey area, so-called 'context' samples.

2.2. Baseline faunal assemblage distribution

A map of baseline faunal assemblage distribution was produced following a similar approach to that set out in Cooper (2012). However, the approach taken in the present study differed in two respects. Firstly, colonial taxa were included in the faunal dataset due to their local importance. The influence of colonial and rarer taxa in subsequent data analysis was assured by initially subjecting data to a fourth-root transformation (see Clarke and Green, 1988). Secondly, clustering of the benthic dataset was performed in R (R Development Core Team, 2010) using the k-means R function available from the *flexclust* library. The k-means method works by finding a solution that minimises the within cluster sum of squares for the *i*th species, summed over all species. The Hartigan and Wong (1979) algorithm was used to find solutions based on different numbers of pre-defined cluster groups. Maps were produced of faunal assemblage distribution based on different numbers of cluster groups. A decision was made as to the appropriate number of

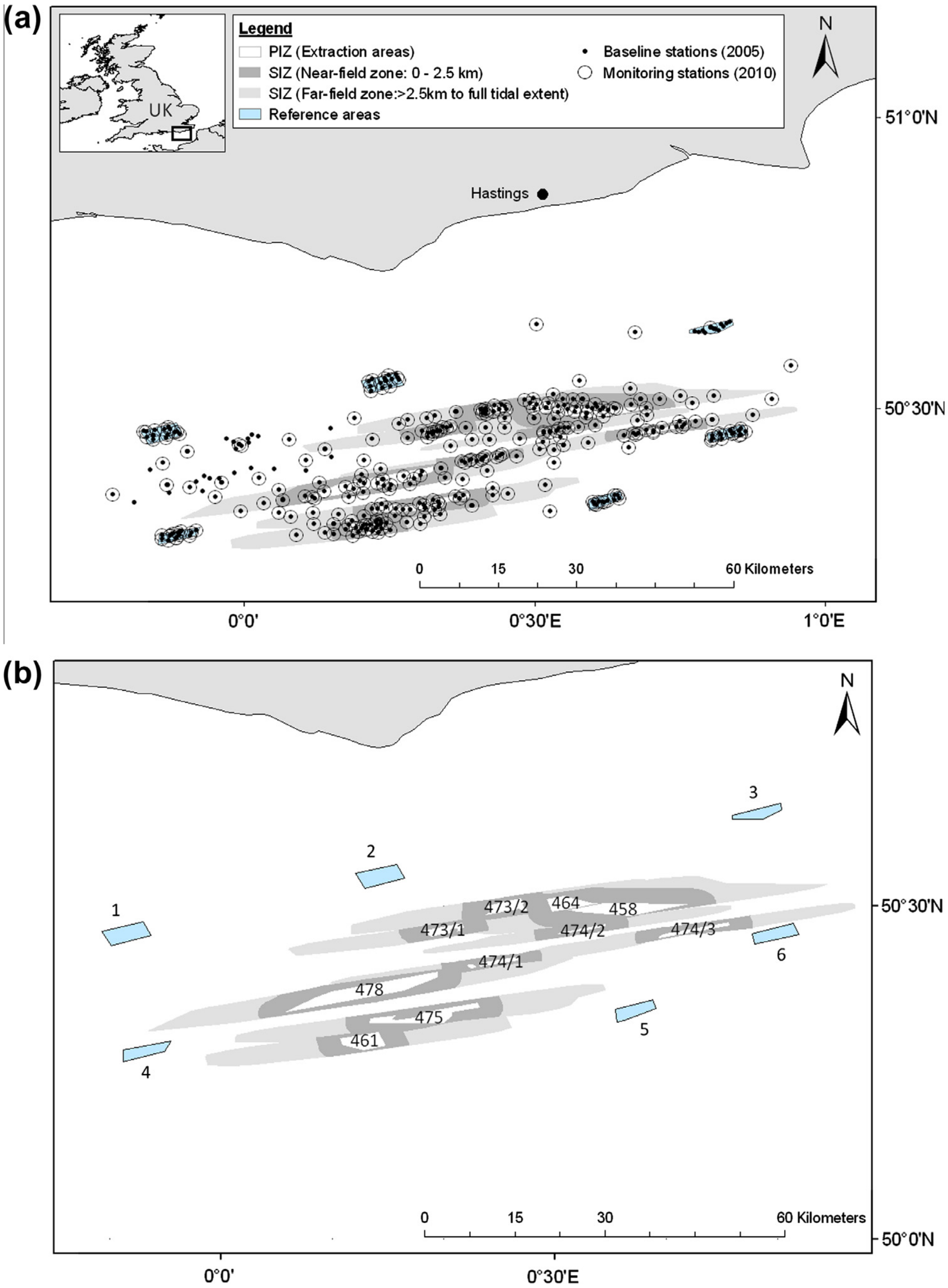

Fig. 1. (a) Sampled locations from the baseline (2005) and monitoring (2010) surveys. (b) Individual licensed extraction areas (numbered white polygons), their associated secondary impact zones, and reference areas (numbered blue boxes). The secondary impact zones are sub-divided into *near-field* (dark grey polygons), and *far-field* (light grey polygons) areas. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.).

cluster groups based on a desire to maximise the level of ecological information, whilst ensuring a sufficient number of sample replicates for subsequent data analysis.

2.3. Baseline faunal assemblage characteristics

The number of taxa (including colonials) and the number of individuals was determined for each macrofaunal sample. These data were used to calculate mean values for each faunal assemblage. A bar chart, showing 95% confidence intervals, was used to examine the difference in both metrics between the different assemblages. The SIMPER routine in Primer (Clarke and Warwick, 1994) was used to identify the characterising taxa from each assemblage.

2.4. Baseline sediment characteristics

Plots of sediment particle size distribution (cumulative weight by sediment size class) were used to compare the sediment composition of samples belonging to each of the identified faunal assemblages. For each assemblage, the mean and upper and lower limits of the cumulative distribution were also plotted. The upper and lower limits, also termed the 'sediment envelope', were simply the highest and lowest values for each sediment size class.

Using the cumulative sediment data, the percentages of major sediment fractions (coarse gravel, medium gravel, fine gravel, coarse sand, medium sand, fine sand, silt/clay) were calculated for individual samples. Sediment fractions were based on the Wentworth classification (Wentworth, 1922). Using these summary data, the mean, and the upper and lower limits were again determined. As before, the upper and lower limits were simply the highest and lowest values for each size class. These values defined the range of sediment composition found in association with each faunal assemblage, and hence the upper and lower limits of acceptable change within extraction areas and their zones of potential secondary effect. Clearly, the full range of sediment composition found in association with each faunal assemblage is more likely to be identified with higher numbers of samples. This fact should provide a powerful incentive to the aggregates industry to acquire more rather than less samples at the baseline characterisation stage.

An ANOSIM test (Clarke and Warwick, 1994) was applied to the same summary dataset to determine whether there were statistically significant differences in the sediment composition of samples belonging to the different faunal assemblages. The *R* value from this test provides a measure of the difference between groups, and would be expected to be in the range from zero to one; a value of zero implies there is no difference between groups, whilst a value of one implies that groups are completely different; an associated *p*-value of < 0.05 was taken to imply statistical significance. The SIMPER routine in Primer was used to identify which sediment fractions accounted for the differences between faunal cluster groups.

2.5. Licence condition

It is proposed that the following standard condition would be applied to all licences:

> *At the end of the licence term, and with allowance made for natural variability, the composition of sediments within the Primary and Secondary Impact Zones must remain within the acceptable change limits for the faunal groups identified during the pre-dredge survey. Compliance will be established using the methodology outlined in this paper.*

The aim of this condition is to ensure that the seabed habitat is maintained in a state that will allow for the return of the pre-dredge faunal distribution after dredging, thus ensuring the long-term sustainability of marine aggregate dredging on seabed macrofaunal communities.

2.6. Survey design

The adequacy of the existing survey design was assessed using statistical power analysis. Specifically, the analysis was used to identify the level of difference in the mean composition of each sediment fraction which might be reliably detected between the baseline and monitoring surveys. This assessment was made for each treatment (e.g. PIZ, SIZ and REF), both at the level of individual extraction site, and using all the data. Analyses were undertaken in Minitab v15 using the Power and Sample size calculator for a two-sample *t*-test. The test required input variables for standard deviation (s), required statistical power ($1-\beta$), and the number of samples available (n). Standard deviation was based on the differences in each sediment fraction between the baseline and monitoring surveys. A power of 0.8 was chosen so that there was a relatively high chance that a difference, if present, would be detected. The number of samples (n) was the number of sites where both a baseline and a monitoring sample were available for comparison.

2.7. Assessing for gross changes in sediment composition

Major changes in sediment composition were identified for all locations (PIZ, SIZ and REF), both at the individual site and meta-analysis level, using a paired sample *t*-test. Tests were performed in Microsoft Excel, with the null hypothesis that the mean (μ) of each sediment fraction was the same before and after dredging. As the difference could be in either direction, a two-tailed test was applied. A *p*-value of < 0.05 was taken to indicate a potential statistically significant difference in the means of the two groups, leading to a rejection of H_0.

2.8. Assessing compliance with the licence condition for acceptable change

A series of line charts, one for each of the identified faunal cluster groups, was produced for the PIZ and SIZ of each extraction area and the Reference sites. The line charts showed the major sediment fractions (coarse gravel, medium gravel, fine gravel, coarse sand, medium sand, fine sand, silt/clay) along the *x*-axis, and percentage contribution along the *y*-axis. Onto these charts were plotted the relevant upper and lower acceptable change limits (see Section 2.4), and the individual monitoring samples data. Where the value of a sediment fraction for any individual sample fell outside the upper or lower limits, this was termed a ‘deviation’. Compliance with the stated licence condition for acceptable change was established for both PIZ and SIZ by comparing the total number of deviations versus the total number of possible deviations (see equation below). The number of possible deviations is simply the number of samples (n) multiplied by the number of sediment fractions (i.e. seven).

$$\%\ \text{Complience} = 100 - \left(\frac{\text{No. of observed deviations}}{\text{No. of possible deviations}} \times 100\right) \quad (1)$$

Changes in either zone (PIZ/SIZ) were deemed acceptable where the percentage compliance was within the range seen for individual reference sites. Where the percentage compliance was less than that observed for any of the reference sites, the site/zone was deemed non-compliant. Analyses were undertaken to assess for change and compliance at individual sites and, using data pooled by treatment categories (PIZ, SIZ and Reference), for the region as a whole.

2.9. Addressing non-compliance

Where a site was deemed to be non-compliant, three further steps followed. Firstly, the likely consequences of the deviations were considered (i.e. what eventual changes in faunal group might result from the altered sediment composition?). This assessment was made using a further line chart showing the individual sample deviations and the upper and lower limits of acceptable change in sediment composition for all faunal cluster groups. On this chart, individual sample deviations were identified using a cross symbol coloured according to the original baseline faunal group. Next, the location of the sample(s) where problem deviations occurred was identified. This allowed the spatial scale of the problem to be observed. Finally, an appropriate management response was identified. What constitutes an appropriate response will vary, but options include the following: do nothing (where natural recovery is expected), reduce extraction rate, target extraction of the problem sediments, change screening practices.

3. Results

3.1. Baseline faunal assemblage distribution

A cluster iteration based on four different faunal assemblages (A–D) was taken forward in the subsequent analysis. Whilst there was clearly some overlap in the distribution of these assemblages, there was a clear transition from group A to group D moving from the west-south-west to the east-north-east of the survey area (Fig. 2). With the exception of the PIZ of Area 474/3, the PIZ and SIZ of all extraction sites and reference areas contained at least two different faunal assemblages.

3.2. Baseline faunal assemblage characteristics

3.2.1. Univariate summary measures

A comparison of the mean number of taxa and individuals revealed clear differences amongst the different faunal assemblages (Fig. 3). The lowest mean number of taxa and individuals was associated with samples from faunal assemblage D. In comparison, cluster groups A, B and C had much higher values of both measures. Of these three groups, faunal assemblage B had the highest values, with similar levels of both measures seen for assemblages A and C.

3.2.2. Species composition

Values of Bray–Curtis similarity showed that samples associated with faunal assemblage D were quite different to all other groups (values were 23% for B, 26% for A, and 28% for C). Values of similarity between groups A, B and C were much higher at ~50%, particularly for groups A and B.

The results of a SIMPER analysis (Table 1) revealed that certain taxa were characteristic of all faunal cluster groups. These taxa included the ribbon worm NEMERTEA, the polychaete *Aonides paucibranchiata*, and the bryozoan *Chorizopora brongniarti*. However, differences were also apparent between the different assemblages. For example, A and B were dominated by taxa that are typical of gravel-rich sediments. These include the crustaceans *Galathea intermedia* (squat lobster) and *Apherusa bispinosa*, polychaetes *Pomatoceros* spp. (Keel worm) and *Laonice bahusiensis,* and the echinoderm *Amphipholis squamata*. A comparison of the species found in association with both these groups suggests that A could be regarded as a slightly more impoverished version of B. Whilst sharing some of the species typically associated with coarser sediments, cluster group C also included species more typical of sandier sediments: for example, the echinoderm *Echinocyamus pusillus*, and the polychaete *Glycera lapidum (agg)*. In addition to the ubiquitous taxa, cluster group D included species typical of sandy sediments: for example, the echinoderm *Echinocyamus pusillus*, and the bivalve molluscs *Gari* spp.

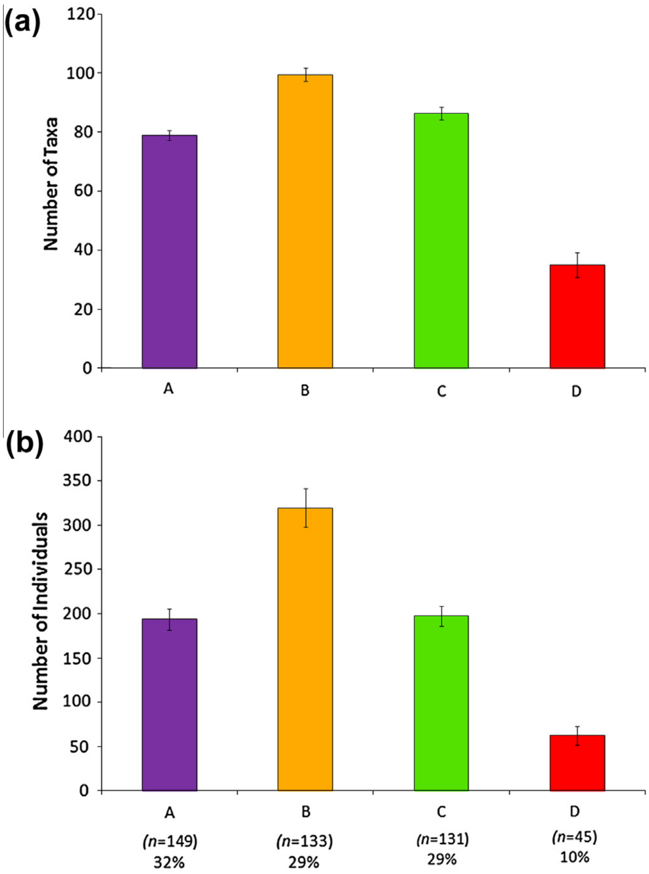

Fig. 3. Mean and 95% confidence intervals for: (a) number of taxa (colonial taxa included), and (b) number of individuals for faunal assemblages A–D. Figures below the bottom graph show the number and proportion of samples belonging to each assemblage.

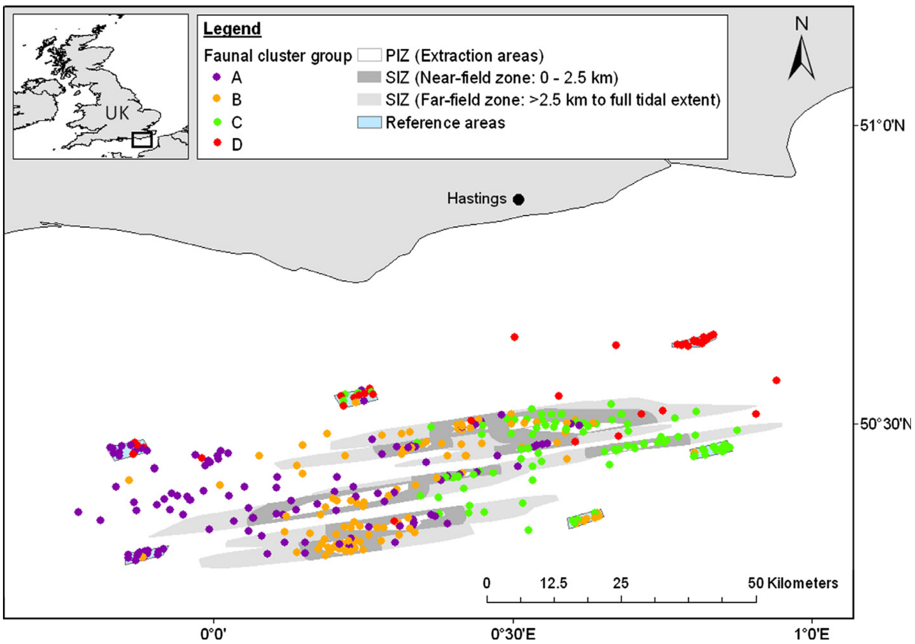

Fig. 2. Distribution of faunal assemblages (A–D) identified through a cluster analysis of the 2005 baseline macrofaunal dataset.

3.3. Baseline sediment characteristics

The cumulative sediment distribution plots show some obvious differences in the composition of samples belonging to the different faunal assemblage groups (Fig. 4). For example, gravel makes up a significant component of the sediment composition of samples associated with groups A and B. The majority of these samples would, according to the Folk classification (Long, 2006), be described as sandy gravels. These assemblages account for the majority (61%) of baseline samples. In contrast, sand was the dominant sediment fraction associated with group D. These samples account for 10% of the baseline samples, and included gravelly sands, slightly gravelly sands and sands. In addition, there were a small number of sandy gravels belonging to this group. Samples belonging to group C account for 29% of the baseline samples, and included similar numbers of sandy gravels and gravelly sands. The proportion of silt/clay in all assemblage groups was generally low, around 1% or less, although higher amounts were occasionally found in association with groups A, B and D. For each assemblage group, the inset tables in Fig. 4 give the mean and upper and lower upper limits for the sediment distribution based on major sediment classes. These limits define the known and therefore acceptable range of sediment composition for each assemblage.

Sediments associated with assemblage D were quite different ($R > 0.5$, $p < 0.05$) from all other groups (see Table 2). These differences resulted largely from the higher proportion of medium sand, and lower proportion of coarse sand and gravel fraction compared to the other groups. Differences between sediments from group C with those from groups A and B are explained by the higher proportion of coarse sand found in association with group C. Differences between groups A and B, although minimal, are explained by the higher proportion of coarse gravel for group B.

3.4. Survey design

Power analysis shows that there were differences between sites in terms of the level of difference in mean sediment composition that can be detected (Table 3). These differences are a result of the differences in the number of samples between sites, and differences in the variability of each sediment fraction within individual sites.

3.5. Assessing for gross changes in sediment composition

Meta-analysis, using data from all sites, revealed a statistically significant increase in fine sand and silt/clay, and a decrease in fine gravel within the PIZ treatment. The increase in silt/clay was also observed in the SIZ treatment, combined with a decrease in coarse sand. Meta-analysis suggests that the increase in silt/clay was greater in the *near-field* zone (i.e. within 2.5 km of the licence boundary) (see Table 3). This observation is consistent with an impact associated with dredging.

Inspection of individual licences (Table 3) revealed that the increase in fine sand within the PIZ treatment was restricted to two sites, Area 474/2 and Area 461. However, all sites showed significant increases in the proportion of silt/clay within the PIZ. Statistically significant changes in sediment composition within the SIZ were only observed for four of the seven dredged extraction areas (Areas 474/1, 474/2, 473/2 and 461). The increase in silt/clay observed in two of the five Reference boxes, and the PIZ and SIZ of the non-dredged licences suggests there may be a non-dredging,

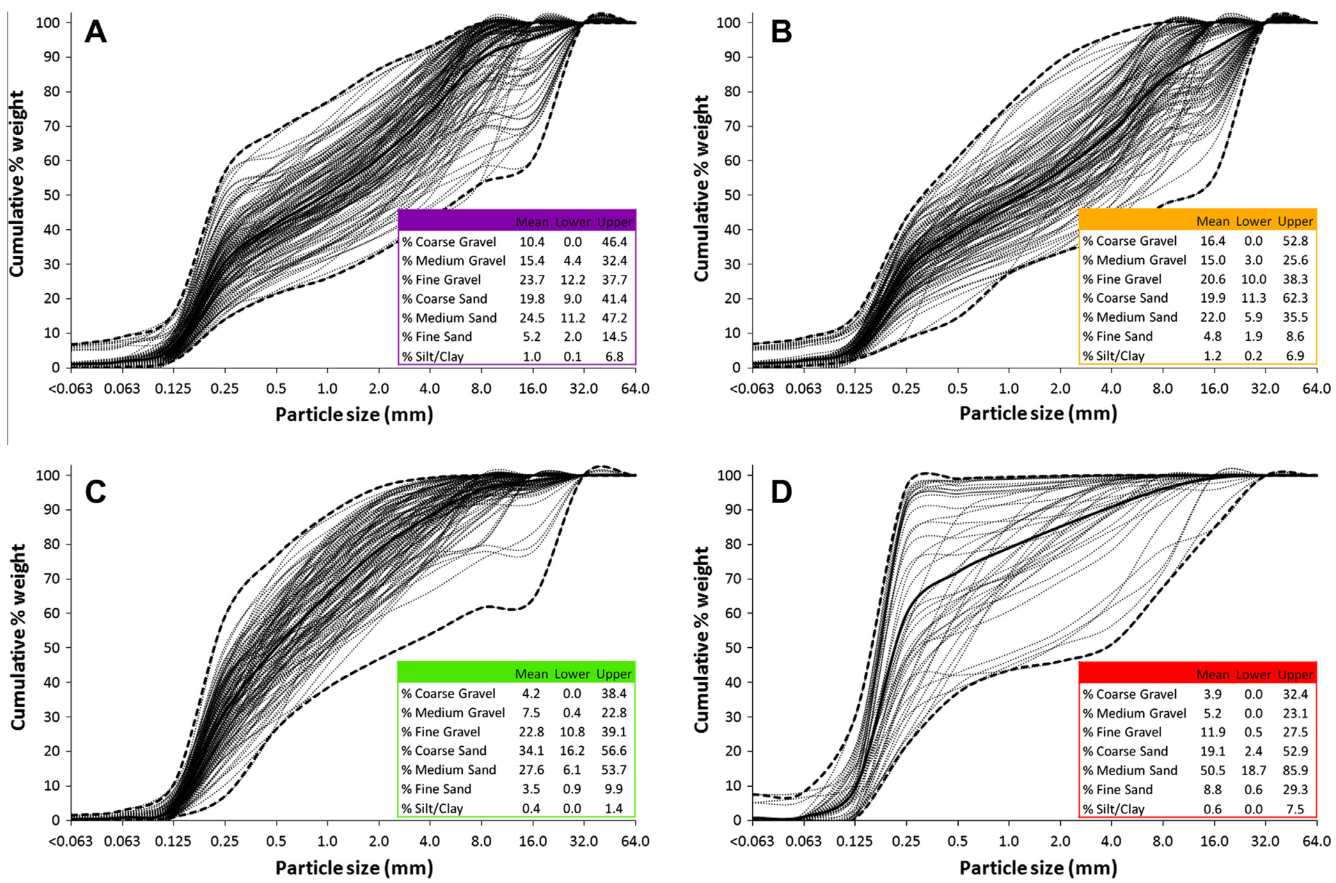

Fig. 4. Cumulative sediment particle size distribution plots for samples belonging to faunal assemblages A–D. The solid and dashed lines show the mean and the upper and lower limits of the distribution. The inset table shows the mean and upper and lower limits of the sediment distribution in terms of percentage contribution of major sediment fractions.

Table 1
Results of a SIMPER analysis showing the characterising species from each faunal assemblage (A–D) accounting for 40% of the within-cluster similarity. Analyses were based on fourth-root transformed macrofaunal abundance data (colonial taxa included).

A	B	C	D
Galathea intermedia	*Galathea intermedia*	*Galathea intermedia*	*Schizomavella sp.*
Apherusa bispinosa	*Pomatoceros*	*Echinocyamus pusillus*	*Echinocyamus pusillus*
Pomatoceros	*Laonice bahusiensis*	*Glycera lapidum (agg)*	NEMERTEA
Laonice bahusiensis	*Pisidia longicornis*	*Gari spp.*	*Eurydice*
Aonides paucibranchiata	*Apherusa bispinosa*	*Aonides paucibranchiata*	*Aonides paucibranchiata*
Amphipholis squamata	*Aonides paucibranchiata*	*Apherusa bispinosa*	*Disporella hispida*
Schizomavella sp.	NEMERTEA	*Typosyllis*	*Puellina spp.*
Rhynchozoon bispinosum	*Harmothoe*	*Atylus vedlomensis*	*Porella concinna*
Microporella ciliate	*Amphipholis squamata*	*Eulalia spp.*	*Escharella ventricosa*
NEMERTEA	*Glycera lapidum (agg)*	NEMERTEA	*Chorizopora brongniarti*
Escharella immerse	*Typosyllis*	*Schizomavella sp.*	*Spio filicornis*
Chorizopora brongniarti	*Epizoanthus couchii*	*Pomatoceros*	
Diastoporidae	*Maera othonis*	*Disporella hispida*	
Porella concinna	*Schizomavella sp.*	*Rhynchozoon bispinosum*	
Glycera lapidum (agg)	*Microporella ciliate*	*Microporella ciliate*	
Ampharete	*Escharella immerse*	*Chorizopora brongniarti*	
Eunice vittata	*Porella concinna*	*Diastoporidae*	
Disporella hispida	*Rhynchozoon bispinosum*	*Reptadeonella violacea*	
Escharella ventricosa	*Reptadeonella violacea*	*Puellina spp.*	
	PORIFERA		
	Cliona (agg)		
	Chorizopora brongniarti		
	Limatula subauriculata		

Table 2
Results of an ANOSIM test based on un-transformed sediment data (% coarse gravel, % medium gravel, % fine gravel, % coarse sand, % medium sand, % fine sand and % silt/clay).

Comparison	*R* Statistic	*p*-Value
D vs B	0.633	0.001
D vs A	0.620	0.001
D vs C	0.542	0.001
C vs B	0.430	0.001
C vs A	0.336	0.001
B vs A	0.065	0.001

possibly natural, component to this increase. Evidence for this is particularly strong at Reference site 1, where residual currents would be expected to take sediment in an east-north-east direction (ECA, 2011). However, the almost universal increase in silt/clay within PIZs where dredging has taken place suggests at least some of the increase is associated with aggregate dredging. No statistically significant changes in sediment composition were found for samples assigned to the 'cumulative' category.

3.6. Assessing compliance with the licence condition for acceptable change

The sediment composition of the majority of monitoring samples fell within the relevant upper and lower acceptable limits, with relatively few deviations (see example area shown in Fig. 5). Overall, values of percentage compliance were within the range seen at reference sites for the PIZ of nine extraction sites, and the SIZ of all ten sites (Table 4). This means that were dredging to have stopped following completion of the monitoring survey in 2010, then these compliant areas should have been able to support a return of the original faunal assemblages, allowing for natural changes. The only site where percentage compliance was outside the reference site values was the PIZ of Area 473/2. The non-compliance at this site is considered further below.

3.7. Addressing non-compliance

3.7.1. Implications

A detailed assessment of the sediment deviations observed within the Area 473/2 PIZ is shown in Fig. 6. Despite taking the sediment composition outside the range for group A (deviations 1, 2, 3 and 5), the altered values remain within the range of acceptability for assemblage group B. As group B is considered a richer version of group A (see Section 3.2.2), such changes could be considered acceptable. In contrast, deviations 4 and 6 take the sediments in the opposite direction (i.e. outside the acceptable limits for group B, but inside for group A). Clearly this is less desirable, but given the similarity between groups A and B, and the localised nature of the deviation, such changes are not of major concern. This leaves deviations 7 and 8. For 7, the higher silt/clay levels are outside the limits for assemblage group C. However, as evidence suggests such changes may be associated with a non-dredging origin, it is sensible to not be too concerned about this. In contrast, deviation 8 takes sediments in the direction of group A to group D. This is of concern as D supports a much reduced faunal assemblage compared to all other assemblage groups.

3.7.2. Location

Deviation 7 occurred at monitoring site 102, located within the eastern end of the PIZ of Area 473/2.

3.7.3. Management action

Research undertaken at 473/2 (ECA, 2011), which acts as a proxy for other licences in the region, has shown that areas of fine sediment accumulation should naturally disperse once dredging ceases, or the rate of extraction drops below a certain threshold level. Given this understanding, the appropriate management response would, at this stage, be to do nothing other than to continue to monitor the situation. Were subsequent monitoring to show that the situation had persisted or worsened then it may be sensible to review this decision.

4. Discussion

4.1. Major findings

In this study, four faunal assemblages were identified within the eastern English Channel. The range of sediment composition found in association with each of these groups was used to define limits for acceptable change in areas influenced by aggregate dredging. Results of a comparison of baseline and monitoring data from 2010 showed evidence of changes in sediment composition,

Table 3
A) Detectable change limits for different sediment fractions (cG = % coarse gravel, mG = % medium gravel, fG = % fine gravel, cS = % coarse sand, mS = % medium sand, fS = % fine sand and S/C = % silt/clay). B) Changes in mean percentage composition of sediment fractions between the baseline (2005) and monitoring (2010) surveys; positive changes are shaded. Statistically significant results, based on a paired 2-sample *t*-test, are underlined.

			A) DETECTABLE DIFFERENCE							B) CHANGE						
ZONE	SITE	*n*	cG	mG	fG	cS	mS	fS	S/C	cG	mG	fG	cS	mS	fS	S/C
PIZ	ALL (Dredged)	89	6.8	2.6	2.0	3.0	3.0	1.0	0.6	+0.3	+0.03	-1.8	-1.1	+0.3	+0.8	+1.4
	473/1	10	12.1	10.5	5.3	5.3	7.7	1.5	2.4	+2.3	-2.4	-3.1	-0.7	+2.0	+0.6	+1.2
	474/2	8	27.9	8.7	5.0	8.1	10.5	2.7	1.8	-3.8	+2.4	-1.8	-0.4	+0.7	+1.6	+1.4
	473/2	11	26.5	4.0	6.2	6.4	12.1	5.4	1.4	-4.0	-1.2	-0.9	+0.2	+2.2	+2.4	+1.3
	461	17	16.4	5.0	5.5	8.6	5.2	2.3	1.9	-5.6	+1.2	-1.1	+1.5	+1.6	+1.1	+1.2
	478	14	18.6	6.2	5.9	4.6	8.6	2.2	1.3	-0.9	+0.4	-2.0	-0.6	+0.6	+0.3	+2.2
	475	16	13.2	7.6	4.5	6.7	4.6	1.2	1.4	+5.8	-0.8	-2.0	-2.9	-1.7	+0.1	+1.5
	458	13	17.9	8.5	6.1	11.6	9.9	2.5	1.1	+6.8	+1.4	-1.9	-4.8	-2.6	+0.2	+0.9
	464 (Un-dredged)	6	12.9	18.1	7.2	12.6	6.6	2.5	1.6	-5.4	+4.8	-0.1	-1.5	+0.2	+0.4	+1.8
	474/1 (Un-dredged)	11	22.0	5.4	5.7	7.5	10.3	2.3	1.0	+10.1	-2.4	-4.8	-6.0	+0.6	+0.9	+1.6
	474/3 (Un-dredged)	9	8.6	5.1	7.3	7.9	7.2	1.5	1.3	+2.7	-1.0	-0.7	-1.8	-0.1	+0.3	+0.6
SIZ	ALL (Dredged)	65	6.2	3.4	2.7	2.5	3.5	0.9	0.7	+2.3	-0.5	-1.0	-1.4	-0.6	+0.0	+1.1
	ALL (Dredged/Near)	32	10.3	4.9	3.8	3.4	5.3	1.4	0.9	+3.8	-0.2	-1.2	-2.4	-1.6	+0.2	+1.5
	ALL (Dredged/Far)	33	7.3	4.8	4.0	3.6	4.7	1.2	1.2	+0.9	-0.8	-0.8	+0.4	+0.4	-0.1	+0.8
	ALL (Dredged/Cumulative)	7	23.8	16.0	8.8	6.2	5.9	2.8	3.1	+7.8	-0.2	-2.4	-3.4	-1.8	-0.5	+0.5
	473/1	8	14.6	10.1	8.3	6.0	8.6	3.0	2.7	+7.9	-1.8	+0.1	-3.1	-3.4	-0.6	+0.8
	474/2	7	33.6	9.6	9.1	10.3	22.5	4.1	2.0	-0.7	-4.4	+2.8	+1.6	-1.6	+0.7	+1.6
	473/2	11	11.9	5.5	7.2	5.5	4.4	1.5	1.3	-0.5	+3.6	+1.3	-2.1	-3.3	+0.0	+0.8
	461	15	14.9	8.4	4.7	4.5	4.2	1.4	0.8	+5.9	+0.2	-1.5	-3.9	-2.0	-0.1	+1.3
	478	15	9.1	8.5	5.2	6.9	7.2	1.5	2.3	+3.6	+0.2	-3.3	-2.6	+0.4	+0.4	+1.3
	475	15	15.8	7.8	6.6	3.7	6.1	1.4	1.5	+1.6	-0.7	-0.6	-1.1	+0.4	-0.3	+0.7
	458	13	12.3	6.5	7.2	7.1	8.8	2.7	1.9	+2.3	+0.4	-2.6	-0.5	-0.4	-0.2	+0.9
	464 (Un-dredged)	10	11.3	4.2	8.7	6.4	6.2	1.9	1.5	-0.7	+2.1	+3.6	-2.0	-3.7	-0.04	+0.8
	474/1 (Un-dredged)	8	21.4	10.7	8.7	9.9	7.4	1.8	1.7	+3.6	+3.7	-1.0	-2.8	-4.7	-0.1	+1.4
	474/3 (Un-dredged)	10	12.3	2.8	6.2	7.6	8.1	1.7	1.3	-1.8	-0.3	+0.5	+1.2	-0.4	+0.1	+0.7
REF	ALL	66	4.5	2.8	3.6	3.1	3.9	1.0	0.9	+1.4	-0.5	-1.5	-0.8	+0.0	+0.18	+1.1
	Context	25	7.9	5.4	5.2	5.1	7.5	2.0	1.1	+1.1	+2.5	-1.1	-0.5	-3.0	-0.5	+1.4
	1	14	5.6	5.5	3.7	1.8	5.5	2.0	1.2	+0.8	+1.8	-0.6	-1.4	-2.8	+0.2	+2.1
	2	13	12.9	6.5	9.2	9.4	11.3	3.4	2.3	+2.7	-1.5	-2.8	+1.4	+0.6	-0.4	+0.1
	4	14	14.3	6.9	8.9	6.2	7.3	1.0	2.4	+1.6	-3.2	-2.5	-0.2	+2.1	+0.03	+2.1
	5	10	13.1	11.3	7.8	13.0	8.2	2.9	2.4	+0.1	+1.1	-0.7	-2.0	-0.4	+0.9	+0.9
	6	14	5.8	3.3	11.0	6.5	12.1	1.8	1.0	+1.7	-0.1	-0.9	-1.9	+0.4	+0.5	+0.4

both as a result of dredging and non-dredging related factors. Despite the changes, the composition of sediment within the PIZ and SIZ of all sites, with exception of the PIZ of Area 473/2, remained within the acceptable limits. A detailed assessment of the sediment deviations at Area 473/2 suggests the problem was confined to elevated levels of fine sand at one station. Given the transient nature of such features (see below), no further intervention was deemed necessary, other than for continued monitoring.

4.2. Importance of findings

The results of this study are important as they suggest that were dredging to have ceased immediately after the 2010 monitoring survey, the composition of sediments would not, in all but the PIZ of Area 473/2, have presented a barrier to full benthic faunal recovery. In addition, the approach to assessing change makes it possible to differentiate between statistically and ecologically significant change in sediment composition, without the need for subjective expert judgement.

4.3. Comparison with other studies

A number of aspects of the methodology employed in the present study differ from those used in the initial trial of the approach (Cooper, 2012), and these differences warrant explanation. Firstly, the epifaunal taxa from grab samples were included in the dataset as a result of their local importance in the eastern English Channel. Secondly, the k-means clustering method was employed here to allow the number of faunal cluster groups to be specified. This was necessary as the initial group average clustering of data produced an unworkable number of statistically distinct cluster groups. Thirdly, the gravel fraction was split into coarse, medium and fine fractions. This was considered necessary given the importance of larger gravel fractions for attachment of certain species. Fourthly, the upper and lower acceptable change limits for different sediment fractions were based on the full range of values rather than a 95th percentile range as used at Hastings (Cooper, 2012). This was necessary as some of the more extreme values, which would have been excluded with a 95th percentile range, occurred within primary or secondary impact zones. Exclusion of these values from the acceptable range would have resulted in deviations even before dredging had begun; clearly this would make no sense. Finally, the approach taken for assessing compliance has changed. In the Hastings study (Cooper, 2012), compliance was established where the mean level of major sediment fractions was within the range defined by the upper and lower limits of sediment composition for the relevant faunal assemblage in the wider region. This approach has two problems. Firstly, it relies on there being an adequate number of replicates with which to assess change. Whilst this was true of the Hastings site, the present study identified multiple faunal assemblages within the PIZ and SIZ of the licensed areas. For some of these assemblages, there was little or no replication. Secondly, it is possible to imagine a situation where values of some sediment fractions might fall outside the acceptable limits, yet the mean value remains unchanged. For these reasons, a different approach to assessing compliance based on changes in sediment composition at individual monitoring stations was used. A consideration of changes in sediment composition at individual monitoring stations has the advantage of allowing the spatial extent of problem areas to be identified, allowing for targeted management intervention.

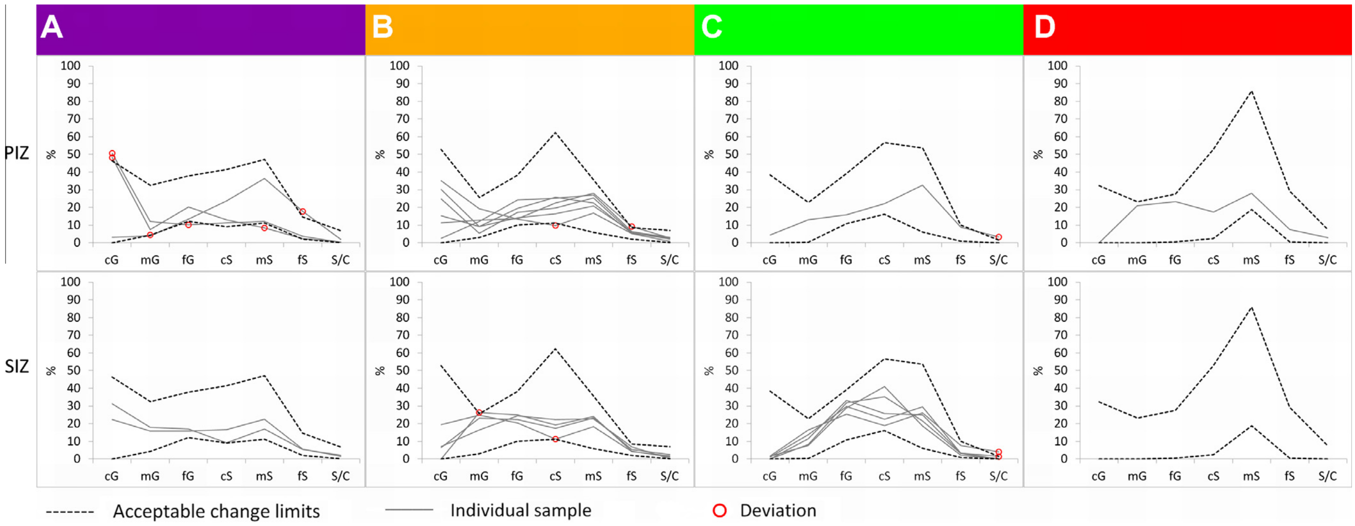

Fig. 5. Example line charts showing the sediment composition of the 2010 monitoring samples in relation to the relevant upper and lower limits for acceptable change. Deviations, where the value of a sediment fraction for any one sample is outside the acceptable change limits, are identified as red circles. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.).

The results of this study are broadly consistent with the findings of the Eastern Channel Regional Monitoring Programme (ECRMP) (ECA, 2011). Included in the ECRMP has been an ongoing assessment of environmental change at Area 473/2, employing a range of techniques to investigate seabed condition (e.g. bathymetry, sidescan sonar, grab sampling for macrofauna and sediment particle size, photography, and sediment tracers). The extent of dredge plumes and the nature of overspill and screened material have also been assessed at this site (HR Wallingford, 2011). Area 473/2 was specifically chosen for such detailed investigation as it was considered to be representative of all extraction sites in the eastern English Channel. In addition, it was where the most intensive dredging was planned, and hence where impacts were most likely to develop. Surveys undertaken in 2007 and 2008 showed that, initially, impacts on sediment and macrofauna were confined to the Active Dredge Zone (ADZ). However, after 2 years of dredging, and following an increase in dredging intensity and the frequency of screening, impacts became apparent in the SIZ. These impacts, evident from sidescan sonar data, comprised of a thin veneer of fine sediment accumulation to the east-north-east of the ADZ and, to a lesser extent, to the south-west of the ADZ. The maximum extent of the feature extended to 2.8 km east-north-east of the licence boundary (ECA, 2011). Whilst the impact features were associated with impacts on the macrofauna, such changes were not detected elsewhere in the SIZ. Following a reduction in the intensity of dredging, these features were shown to dissipate under the action of tidal currents, and in 2010 they had disappeared altogether. The transient nature of impacts on the benthic fauna within the SIZ in this location is an important finding.

4.4. Alternative explanations

Given these results, it is possible that the limited evidence for change in sediment composition within the SIZs from the present study may have resulted from: (1) the localised nature of secondary effects (see ECA, 2011), and the likelihood that the existing monitoring stations were not located within impacted areas (for this reason it is essential for this area to also undertake some acoustic monitoring of PIZs and SIZs) and (2) The possibility that some stations within the SIZ should, on the basis of the evidence presented in ECA (2011), have been assigned to the context treatment – where stations are assigned to the wrong treatment this can have the effect of masking change.

4.5. Limitations of approach

The need to specify the number of faunal cluster groups inevitably introduces a subjective element into the process, and some judgement needs to be exercised in this regard. The judgement has to balance the need to preserve the detail (i.e. maximise the number of faunal groups), whilst at the same time seeking to ensure that the likely full range of sediment composition is effectively identified for each group. As such, it may be sensible to opt for more cluster groups where the number of available samples and the spatial extent of their coverage is large, and to accept fewer cluster groups where the number of samples, or the spatial extent of their coverage is lower. For the sake of consistency, it may also be sensible for such assessments to be made by the same individual.

In contrast to the Hastings study site (Cooper, 2012; Cooper et al., 2007), very little information exists concerning the potential for faunal recovery in the eastern English Channel dredging region. As a result, the prediction that a full recovery will be possible, assuming sediments remain within the defined acceptable limits, remains a hypothesis which needs to be tested. Were sites shown not to recover, despite complying with the licence condition for acceptable change in sediment composition, then it may be necessary to adopt more a more conservative range of acceptable sediment conditions.

4.6. Implications

Given the success of the Eastern Channel Regional Monitoring Programme (ECA, 2011) in informing the understanding of dredging impacts in the region, it is sensible that the developer (Eastern Channel Association) wishes to review the existing monitoring arrangements for the next 5 years (2011–2016), to ensure that the level of effort remains proportionate to both the understanding of impacts and the level of environmental risk (ECA, 2011). From the perspective of the industry regulators, the results of the ECRMP are likely to provide confidence that the environmental impacts from dredging in the ECR are conforming to, or are less than, what was predicted in the Environmental Statements. However, the difficult issue for the regulator, in this and all extraction areas, remains one of being able to identify the point at which impacts become unacceptable. The findings of the present study have the potential to offer benefits here to both parties. For

Table 4
Compliance table detailing the number of samples (n), number of possible deviations ($n \times$ number of sediment categories (7)), number of observed deviations, and overall compliance. The percentage compliance is 100 − ((Number of observed breeches/Number of possible breeches) $*$ 100).

		NUMBER OF SAMPLES (n)					POSSIBLE DEVIATIONS					OBSERVED DEVIATIONS					COMPLIANCE	
ZONE	SITE	A	B	C	D	Total	A	B	C	D	Total	A	B	C	D	Total	%	Acceptability
PIZ	ALL (Dredged)	26	46	15	2	89	182	322	105	14	623	14	12	14	0	40	94	✓
	473/1	4	5	1	0	10	28	35	7	0	70	0	0	1	0	1	99	✓
	474/2	4	2	2	0	8	28	14	14	0	56	1	1	3	0	5	91	✓
	473/2	3	6	1	1	11	21	42	7	7	77	6	2	1	0	9	88	x
	461	3	14	0	0	17	21	98	0	0	119	1	7	0	0	8	93	✓
	478	5	9	0	0	14	35	63	0	0	98	5	1	0	0	6	94	✓
	475	6	8	1	1	16	42	56	7	7	112	1	1	1	0	3	97	✓
	458	1	2	10	0	13	7	14	70	0	91	0	0	9	0	9	90	✓
	464 (Un-dredged)	1	2	3	0	6	7	14	21	0	42	0	0	4	0	4	90	✓
	474/1 (Un-dredged)	4	2	5	0	11	28	14	35	0	77	2	0	4	0	6	92	✓
	474/3 (Un-dredged)	0	0	9	0	9	0	0	63	0	63	0	0	5	0	5	92	✓
SIZ	ALL (Dredged)	18	29	14	4	65	126	203	98	28	455	1	4	7	4	16	96	✓
	ALL (Dredged/Near)	9	16	5	2	32	63	112	35	14	224	1	3	6	1	11	95	✓
	ALL (Dredged/Far)	9	13	9	2	33	63	91	63	14	231	0	1	1	3	5	98	✓
	ALL (Dredged/Cumulative)	1	5	1	0	7	7	35	7	0	49	0	1	1	0	2	96	✓
	473/1	1	5	2	0	8	7	35	14	0	56	0	0	1	0	1	98	✓
	474/2	2	2	1	2	7	14	14	7	14	49	0	0	0	2	2	96	✓
	473/2	2	4	5	0	11	14	28	35	0	77	0	2	2	0	4	95	✓
	461	4	11	0	0	15	28	77	0	0	105	0	1	0	0	1	99	✓
	478	9	3	3	0	15	63	21	21	0	105	0	0	2	0	2	98	✓
	475	3	8	4	0	15	21	56	28	0	105	0	3	0	0	3	97	✓
	458	0	5	6	2	13	0	35	42	14	91	0	0	6	2	8	91	✓
	464 (Un-dredged)	1	3	6	0	10	7	21	42	0	70	0	1	2	0	3	96	✓
	474/1 (Un-dredged)	3	2	3	0	8	21	14	21	0	56	0	1	3	0	4	93	✓
	474/3 (Un-dredged)	1	1	8	0	10	7	7	56	0	70	0	1	3	0	4	94	✓
REF	ALL	25	9	21	11	66	175	63	147	77	462	2	7	10	25	44	90	n/a
	Context	11	4	5	5	25	77	28	35	35	175	1	3	4	1	9	95	n/a
	1	10	0	0	4	14	70	0	0	28	98	0	0	0	0	0	100	n/a
	2	2	1	4	6	13	14	7	28	42	91	0	3	3	4	10	89	n/a
	4	13	1	0	0	14	91	7	0	0	98	2	0	0	0	2	98	n/a
	5	0	6	4	0	10	0	42	28	0	70	0	4	1	0	5	93	n/a
	6	0	1	13	0	14	0	7	91	0	98	0	0	6	0	6	94	n/a

example, the ability to assess a change in relation to pre-defined limits will allow the regulator, and their scientific advisors, to differentiate between statistical and ecological significance, and hence when management intervention is warranted (see Cooper, 2012). However, it is important to recognise that, in addition to the benthic macrofauna, other factors may influence the limits of acceptable change in sediment composition. For example, the specific requirements of herring spawning grounds (see de Groot, 1980).

For the developer, the method may allow for a more cost effective approach to monitoring given that approximately 90% of the cost of processing grab samples is associated with the macrofauna, with 10% for the sediments (assuming respective processing costs of £450 and £50 per sample). There are also savings in terms of reporting, as the purpose of monitoring within the PIZ and SIZ would simply be to determine compliance with the stated licence condition(s) for sediment particle size. As a result, reports will be shorter, making them quicker to produce and, ultimately, to assess.

4.7. Adoption of approach

Were a decision to be made to switch to this approach to monitoring, it is recommended that the detailed programme of investigation at Area 473/2 is continued in order to address outstanding questions concerning recovery, both within areas subject to the direct effects of dredging, and in the SIZ where fine sediments have been deposited and subsequently dissipated. In addition, it is recommended that monitoring of sediments and macrofauna within reference boxes and context stations is also continued. This work will allow the broadscale characterisation of region to be kept up-to-date, reducing the need for additional characterisation surveys in support of new licence area applications. In addition, it will provide a time series for macrofauna and sediments in the region. Analysis of trends may help to identify if the capacity of the environment to cope with dredging, and other anthropogenic pressures in the region, is exceeded (see Barrio Froján et al., 2008). The data could also usefully contribute to UK monitoring programmes. Asking the aggregates industry to contribute, indirectly, to such initiatives has some logic given that their activities, in combination with other anthropogenic pressures, may have a bearing on the status of the UK seas. The time-series would also provide a check on the health of surrounding faunal assemblages. This is important as these areas will have an important role, through provision of individuals and larvae, in the eventual recolonisation of impacted areas.

Adoption of the monitoring approach used in this study for other dredging regions in the UK is considered achievable, although some work would be required to establish appropriate baseline conditions using a combination of historic and new samples. To determine what new sampling will be required, a regular triangular grid of stations will need to be established within the PIZ and SIZ of each extraction site. The use of a regular grid will allow for better spatial coverage of the zone, resulting in an improved ability to detect changes should they occur (see Barry and Nicholson, 1993). The required number of samples, and hence size of the grids, will be informed by the need to achieve adequate spatial coverage of each zone. In addition, power analysis can be used to try to ensure that the number of collected samples will allow for parity of detection differences between sites, both within and between regions. Within these grids, existing benthic data

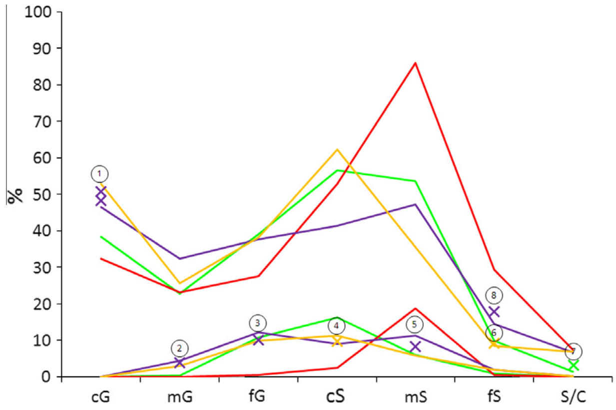

Fig. 6. Line chart showing deviations (crosses) and the acceptable change limits in sediment composition for all four assemblage groups (lines). Lines and crosses are coloured according to the relevant cluster group (i.e. A – purple, B – orange, C – green, D – red). Individual deviations are numbered for the purpose of discussion in the text. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.).

should be assigned to the nearest grid node. Making use of historic sample data in this way will reduce costs for industry by reducing the number of new samples that need to be collected. In addition, the historic samples will provide data on an earlier condition of the sampled station. At nodes with no existing data, new data will be needed for macrofauna and sediments. Clearly this will involve an initial up-front cost to industry. However, costs will be similar to those of a typical pre-dredge benthic survey – which will no longer be required. Over the long-term, it is anticipated that industry will make considerable savings through not having to sample macrofauna within the PIZ/SIZ areas.

5. Conclusion

Monitoring data from 2010 showed that the composition of sediments within the impact footprint of nine of the ten extraction areas in the East Channel Region remained within acceptable limits. This means that changes, should they persist, are unlikely to have major long-term ecological significance post-dredging. For Area 473/2, where the level of percentage compliance was below the minimum acceptable level, the changes were judged to not warrant further intervention given the demonstrated likelihood of natural recovery. The approach taken in this study offers a simple, objective and cost-effective means of assessing the acceptability of changes in sediment composition, and could simplify the existing monitoring regime.

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Paper #2

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Can the benefits of physical seabed restoration justify the costs? An assessment of a disused aggregate extraction site off the Thames Estuary, UK

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ABSTRACT

Physical and biological seabed impacts can persist long after the cessation of marine aggregate dredging. Whilst small-scale experimental studies have shown that it may be possible to mitigate such impacts, it is unclear whether the costs of restoration are justified on an industrial scale. Here we explore this question using a case study off the Thames Estuary, UK. By understanding the nature and scale of persistent impacts, we identify possible techniques to restore the physical properties of the seabed, and the costs and the likelihood of success. An analysis of the ecosystem services and goods/benefits produced by the site is used to determine whether intervention is justified. Whilst a comparison of costs and benefits at this site suggests restoration would not be warranted, the analysis is site-specific. We emphasise the need to better define what is, and is not, an acceptable seabed condition post-dredging.

1. Introduction

Several strategic European directives require the EU member states to consider the restoration of impacted environments (e.g. Article 2 of the EEC Habitats Directive (Council Directive 92/43/EEC), the EC Water Framework Directive (Directive 2000/60/EC), the EC Marine Strategy Framework Directive (Directive 2008/56/EC) and Article 2 of Annex V to the OSPAR Convention (OSPAR, 1992)). Studies into seabed recovery following marine aggregate dredging show that physical and biological impacts may persist long after the licence term (Foden et al., 2009). Physical changes to the seabed can broadly be categorised either as 'topographic' or 'sedimentary'. Topographic changes include the creations of dredge furrows and dredge pits. Dredge furrows are created by Trailer dredging and are typically 1–3 m wide and 0.2–0.3 m deep (Kenny and Rees, 1994). Dredge depressions are created by Anchor dredging, and are typically 8–10 m in depth, but can reach 20 m (Dickson and Lee, 1972; Newell et al., 1998; Boyd et al., 2004). Changes in the composition of seabed sediments result from the exposure of different underlying sediments (Kenny and Rees, 1994; Cooper et al., 2007), but are more commonly associated with the practice of sediment screening, whereby unwanted sediment fractions, usually sands, are returned to the seabed (Hitchcock and Drucker, 1996). Depending on local conditions, physical impacts can extend beyond the boundary of the licence area (Newell et al., 2004a). These physical impacts, particularly those associated with altered sediment composition are also likely to affect biological recovery (de Groot, 1986; Boyd et al., 2005; Gray and Elliott, 2009; Cooper et al., 2011b, 2011c).

The period of time such features persist will depend on hydrodynamics, sediment particle size and the intensity of the activity (Foden et al., 2009; Elliott et al., 2007; Borja et al., 2010). This questions the nature and need for a management response which in turn depends on the spatial extent and duration of such physical impacts. The seabed could be left to recover naturally, whilst accepting that some changes in its character are inevitable. Such changes may be acceptable given the relatively small size of affected areas, and as dredging permissions are only granted in areas where the potential impacts of such dredging are deemed to be acceptable. However, when the permission is granted it is not always clear how long any residual effects of dredging may last, and this will influence the overall acceptability of the project. In addition, minor impacts may be in combination and cumulative, although the significance of such changes for the wider ecosystem, which may be defined in statistical, ecological and societal terms

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(Gray and Elliott, 2009; Elliott, 2011), is poorly understood. Developers are required 'to leave the seabed in a similar condition post-dredging' (ODPM, 2002), thus ensuring the sustainability of environmental (seabed) use, central to The Ecosystem Approach (CBD, 2000). Hence there is a need to further understand the significance of the physical impacts of dredging (see Cooper, 2012), and to determine whether there are realistic options for seabed restoration.

Physical restoration could use methods routinely employed during either aggregate or capital/maintenance dredging. These include:

- Dredging unwanted material from the seabed using a conventional Trailer Suction Hopper Dredger (TSHD) (see PIANC, 2009). The dredged material can then be used commercially, placed as infill for dredge depressions, discharged at a recognised disposal site, or used for beach recharge.
- Capping, which is typically used to isolate contaminated sediments from the surrounding environment (e.g. Simpson et al., 2002), may be used to restore habitats, especially restoring sediment character (Rees et al., 2002; Collins and Mallinson, 2006; Ware et al., 2010; Cooper et al., 2011a). The type of TSHD dictates the method and rate of discharge of material.
- Bed levelling using a dredge plough can level high spots which remain after bulk dredging (Bray et al., 1997). Whilst this approach is very effective in the context of maintenance dredging, where water depths are relatively shallow and sediments are comparatively soft, its effectiveness in the typically deeper water and coarser sediments of aggregate extraction areas is largely unproven. Preliminary results from an extraction area in French waters did not show any obvious physical effect resulting from levelling using a 5 m plough (Dr. Michel Desprez, Université de Rouen, France, pers. comm.).
- In deep offshore conditions, bed levelling can be undertaken using a hopper dredger (Jean-Baptiste De Cuyper, Dredging International, pers. comm.) as seen in offshore pipe-laying, where a TSHD draghead is moved over the seabed, while using its jets to fluidize the sediment. No actual dredging of material occurs and if normal jetting is not adequate then reverse water pumping, via the dredge pumps, could further agitate the seabed and level it after settling. This technique has advantages: a rigid dredge pipe instead of cables gives better control of the draghead/plough at greater depths; the larger dimensions and higher freeboard of TSHD enable working in worse offshore weather conditions; the maximum operating depth is more than 100 m; vessels can be equipped with mapping and survey equipment (often via a moonpool) and are fully equipped to work offshore for longer periods; vessel crews are trained and experienced for dredging (this would not be the case if ploughing technology were mounted on a supply vessel), and has better results in sand and gravel (production rates).

Small-scale restoration experiments have shown that it may be possible to address, to a greater or lesser extent, some of the residual physical impacts left by extraction activities. For example, gravel seeding could be used to restore the composition of seabed sediments in areas characterised by an overburden of sands from sediment screening (Cooper et al., 2011c). Cooper et al. (2001c) also showed that gravel seeding was effective in returning the fauna to a state more similar to the gravely reference site. Collins and Mallinson (2006) considered using waste shell material, resulting from shellfish processing, to help restore the seabed, and to promote benthic faunal recolonisation, particularly of species which require this type of substratum for attachment. Finally, bed levelling has been trialled off the French coast, in an area of seabed characterised by dredge furrows, as a means of enhancing faunal recovery (Dr. Michel Desprez, Université de Rouen, France, pers. comm.). However, it is not yet clear whether the use of such techniques is practicable on an industrial scale, and whether the benefits justify the costs. Bellew and Drabble (2004) discuss the environmental and economic cost of various rehabilitation options against the do-nothing option of natural site recovery for marine aggregate licence areas; their study suggests that dredging clearly has impacts on the environment but remediation can also have impacts and be costly. The Wildlife Trusts (2006) noted that, 'overall, there was near-consensus that active restoration (for example, habitat replacement or creation) is rarely viable in the marine environment...the cost of such intervention is typically prohibitive'.

The present study addresses such issues by examining a site in the Thames Estuary where extraction of aggregates has ceased. We firstly identify areas of persistent physical impact on the seabed at the study site and identify restoration techniques and their cost. Secondly, we consider the significance of the persistent impacts of dredging for the wider ecosystem, based on an assessment of the ecosystem services and goods/benefits provided by the site. Finally we consider whether there is any merit in seeking to restore the seabed at the study site and make recommendations regarding restoration of other extraction sites.

2. Methods

Methods are required to assess whether the benefits of physical seabed restoration following cessation of aggregate dredging justify the costs. An assessment requires the evaluation of benefits against costs incurred, possibly within the formal framework of an economic cost benefit analysis (CBA) for environmental policy purposes (Hanley and Barbier, 2009). Since there are many environmental CBA studies, this paper highlights some application-specific points, beginning with the DPSIR framework (see below) which can identify key elements and their inter-relationships. Establishing a restoration plan includes identifying impact zones and restoration techniques, and its costs include restoration works, licensing, carbon footprint, and survey work. These costs can be set against anticipated benefits of seabed restoration associated with changes in the provision of ecosystem services. Finally, we outline a case study site for an application.

2.1. DPSIR framework

The DPSIR (Drivers-Pressures-State Changes-Impacts-Responses) framework can be used as a problem structuring method for managing the marine environment (Gregory et al., 2013) and can be applied to the specific problem addressed by this paper. The approach is consistent with The Ecosystem Approach, as advocated for example by the Marine Strategy Framework Directive (European Commission, 2008). Following Svarstad et al. (2008), who argue that the boundary of the system that the DPSIR framework describes depends on the issue of interest and its conceptualisation, the key boundary conditions here are those relevant to the marine aggregates sector and a DPSIR framework which summarises components of the sector is depicted in Fig. 1. This depiction allows for the potential for site restoration as a management response. The physical nature of the seabed is affected by the aggregate dredging activity, for example the creation of dredge furrows and depressions and increased sand composition, as well as through restoration measures including gravel seeding, dredging/infill, bed-levelling and capping; the latter measures reflected by the feedback loop. Benthic marine organisms, demersal fish and shellfish species, carbon sequestering organisms and habitats may be affected through both extraction and restoration. Changes in suspended sediment and in Greenhouse Gas (GHG) emissions,

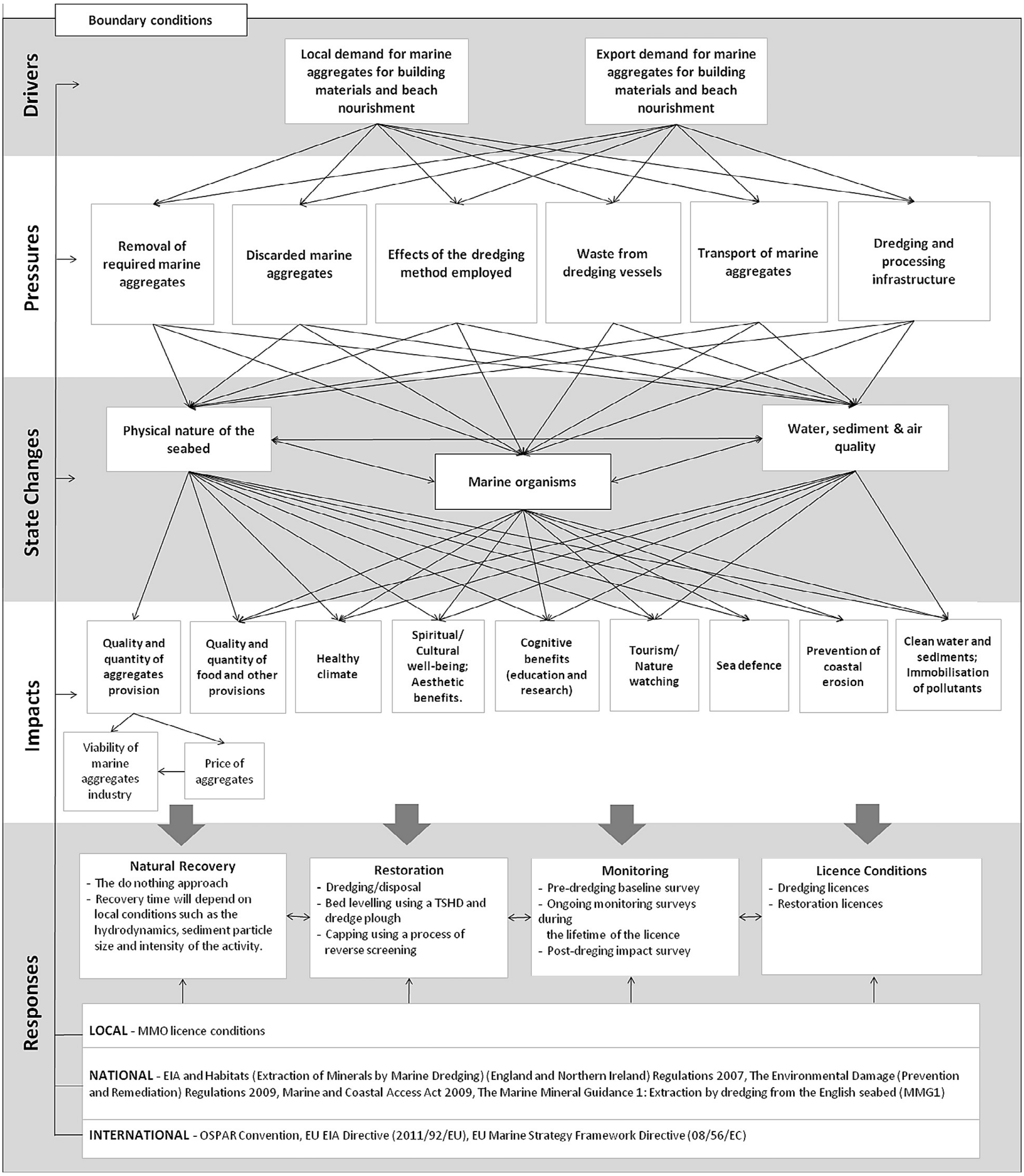

Fig. 1. A DPSIR framework for the management of the UK marine aggregates extraction industry.

and the appearance of surface oils and remobilisation of pollutants are captured within the water, sediment and air quality element. Removal of spawning and nursery habitats and fish-food impact on, for example, food provision from the aggregates site, with restoration expected to mitigate these effects (Fig. 1) thus showing the complexity of the system reflected by this DPSIR.

2.2. Establishment of restoration plan

The choice of restoration technique, identified as a *Response* in Fig. 1, depends on the observed changes in the seabed following cessation of dredging. A combination of sidescan sonar and multibeam bathymetric data is required to identify changes in the sediment composition and topographic features, respectively. Once imported into a GIS software package (such as Mapinfo®) polygons can be drawn around separate impact zones and the area of each feature can be calculated. The extent to which the volume of substratum has changed can be calculated from bathymetry/topography by importing the multibeam data into the software package Fledermus®, and comparing the observed positive or negative shape to a flat surface, modelled by the software, considered to represent the likely pre-dredge surface.

Once impact zones have been identified, a suitable method for physically restoring the seabed within each zone can be identified. Restoration tasks, including survey work, also need to be highlighted within a strategy to maximise the use of local materials and ship time in order to minimise costs.

2.3. Restoration costs

Costs fall into one of four categories: restoration works, licensing, carbon footprint, and survey costs. The cost of restoration works should be calculated for each impact zone, based on its area/volume, and the chosen restoration method (see Cooper et al., 2011a).

Any restoration actions placing material on the seabed require a marine licence whose fee is determined according to the weight of material placed (see http://www.marinemanagement.org.uk/works/licensing/fees_disposal.htm#dredged). Multiple placements of material can be considered as one application as long as each is undertaken as part of the same works campaign, but planned breaks in a campaign incur separate applications fees. The overall cost of the marine licence would be apportioned by zone, in relation to the tonnage of material to be deposited.

With regard to the carbon footprint, at present the marine aggregates industry is not required to participate in any 'cap and trade' system designed to limit GHG emissions. Nevertheless, emissions from restoration represent a cost to the environment, and should therefore be included in assessing the restoration costs and benefits. Defra (2010b) provide a methodology to calculate the traded cost of GHG emissions and the same methodology is followed here.

The survey costs associated with the restoration include a minimum of two surveys plus any normal post-dredge survey. The first survey assesses the significance of impacts and the extent of recovery, provides data to allow development of a detailed restoration plan, and forms a baseline against which the success of restoration may be judged. This survey would include a full coverage acoustic survey of the site, together with ground truthing using 0.1 m^2 Hamon grab, camera and 2 m beam trawl samples. A minimum of 10 sample grab replicates would be required from within each impact zone and reference site (Cooper et al., 2011a). Following restoration works, a 'post-restoration' survey would establish whether the work had been successful. As the restoration aims to address physical changes, this survey would not include a biological component. The cost of survey work includes vessel time, staff time, sample processing, and data analysis/reporting.

2.4. Ecosystem services and goods/benefits

We take the view here that ecosystem services are the means to produce societal benefits and are embedded within the DPSIR framework through their relationship to State changes and Impacts. Since the concept of ecosystem services achieved prominence in the Millennium Ecosystem Assessment (MEA, 2005) a large literature has emerged exploring its definition and evaluating its constituent parts (see for example Beaumont et al., 2007; Fisher et al., 2009; Mace et al., 2011; UK NEA, 2011). Marine ecosystems comprise a range of components (habitats, species, sea space, and others) and processes (production, decomposition, food web dynamics, and others) which lead to the delivery of both intermediate ecosystem services (primary production, nutrient cycling, carbon sequestration, and others) and final ecosystem services (fish and shellfish, genetic resources, climate regulation, and others) (Fig. 2, adapted from Atkins et al., 2011a; Turner et al., unpublished results). Through both intermediate and final services, and the use of complementary human and man-made capital (as inputting skills, time, energy, machinery and equipment, etc.) society secures goods/benefits from marine ecosystems (food, raw materials, medicines, sea defence, tourism, recreation and others). A good/benefit is defined here as something of anthropocentric instrumental value, i.e. of both personal use (direct and indirect) and non-personal use (bequest, altruistic, existence values).

Since ecosystem services have potential to lead to benefits for human wellbeing it is appropriate to consider their value. For some marine ecosystem services market prices may reflect their value, but for others a market price either does not exist or is inadequate. A range of methods is available to assess the values that are placed on these benefits (Table 1) including economic valuation techniques and their relevance to ecosystem services through the goods/benefits that may result. However, it is not appropriate to value basic processes and intermediate services without identifying explicitly the associated final ecosystem services and goods/benefits which have human welfare implications. Many of the methods are categorised as non-market valuation approaches as they do not rely on market prices; such methods are gaining wider acceptance and are advocated by the UK Government for policy evaluations (HM Treasury, 2003). Where there is a market price (e.g. marine aggregates) it might not be appropriate to recommend the use of a non-market approach (such as CVM or CEM) to value the sand and gravel extracted if this is required, whereas it would be appropriate to recommend CVM or CEM to value the impact of aggregate extraction on some other ecosystem services (e.g. tourism and nature watching). In addition, several general matters must be addressed for the effective valuation of marine ecosystem services, including the need to avoid double counting in valuation; spatial explicitness to clarify the level of understanding as ecosystem services are context dependent; marginality associated with the requirement that valuation should focus on incremental changes in ecosystem services rather than larger impacts; non-linearities which refers to the nature of the relationship between a given disturbance and its impact on ecosystem services; and threshold effects where a marginal disturbance can lead to an abrupt change into an alternative state (Turner et al., 2010; Atkins et al., 2011b).

Fig. 1 identifies the key ecosystem services and goods/benefits associated with aggregate extraction. Focussing on aggregate extraction and site restoration, a qualitative assessment of changes in ecosystem service provision can be made, based on evidence from the literature and expert opinion. Where site-specific scientific analytical data are available, a quantitative assessment of changes in ecosystem service provision may be feasible and form the basis for an economic valuation using techniques in Table 1.

2.5. Case study site

Area 222 is an historic aggregate extraction area occupying 0.3383 km^2. The site is located in the outer Thames Estuary, 37 km east of Felixstowe in water depths of 27–35 m (Lowest Astronomical Tide) (Fig. 3; Boyd et al., 2003, 2005, give a detailed description of the site). A total of 10.2 million tonnes of aggregate (sand and gravel) were extracted from the site over a 25 year period (1972–1996), before the site was relinquished in 1997. Material was extracted using both trailer suction and static dredging techniques, and cargoes were screened in order to adjust the ratio of sand to gravel, with rejected material being returned to the seabed (Boyd et al., 2005). Owing to the age of the licence it did not fall within any Environmental Impact Assessment (EIA) requirements, under the European EIA Directive (2011/92/EU), and did not have any associated environmental monitoring conditions, except the need to conduct regular bathymetric surveys (S. Gibson, Royal Haskoning, Redhill, UK, pers. comm.). However, the site's post-extraction status was extensively researched between 2000 and 2007, 4–10 years after dredging (Boyd et al., 2003; Boyd et al.,

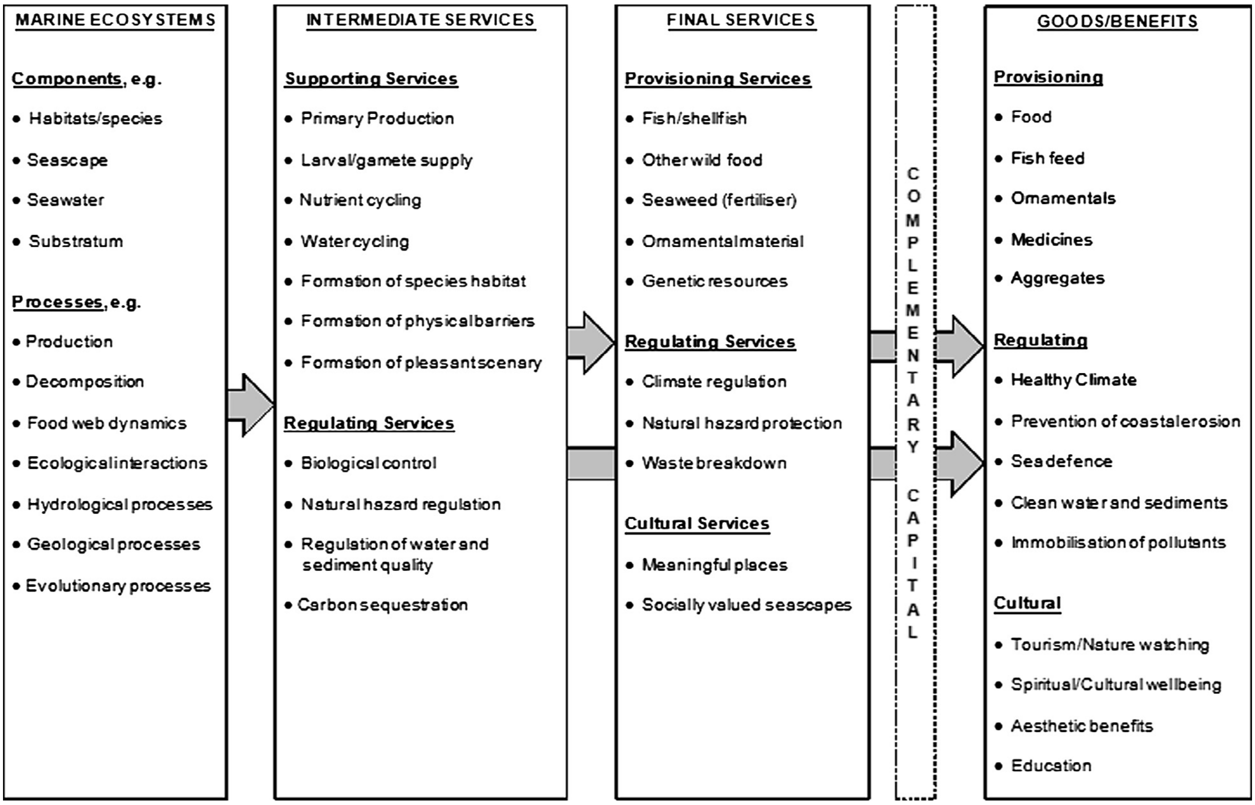

Fig. 2. An ecosystem services framework for the marine environment (adapted from Atkins et al., 2011a; Turner et al., unpublished results).

2004; Boyd et al., 2005; Cooper et al., 2005; Wan Hussin et al., 2012). This showed that physical and biological recovery occurred in areas of relatively low dredging intensity after 7 years. In contrast, recovery within areas subjected to higher dredging intensity appears to be ongoing, with recovery predicted to take at least 15–20 years (Cooper et al., 2011a).

3. Results

3.1. Establishment of a restoration plan

The most recent sidescan sonar and multibeam bathymetric data from 2004, 8 years after cessation of dredging at Area 222 (Cooper et al., 2005), were imported into the GIS software package Mapinfo®. Seven zones of probable impact resulting from marine aggregate dredging were identified from a detailed assessment of the sidescan sonar backscatter and multibeam bathymetric data. Zones 1–4 are associated with topographic changes to the seabed, and occupy an area of 0.839 km^2 (Fig. 4a). Zones 5–7 are associated with changes in sediment composition, and occupy an area of 0.743 km^2 (Fig. 4b). An area of intersection between zones of topographic and sediment changes amounts to 0.158 km^2 (Table 2).

Zone 1 is characterised by dredge depressions, some partially infilled with sand. Zone 2 is an elongated depression located to the east of the licence area and running northwest towards zone 1. Zone 3 largely coincides with the licence area and is characterised by an irregular seabed, with some evidence of trailer dredging activity. Zone 4 comprises a series of sandwaves located on the southern boundary of zone 1. Zone 5 comprises areas of rippled sand. Some patches associated with zone 5 sit within zone 6, an area of sand ribbon/streaks. Zone 7 comprises areas of rippled sand to the north and south of the site. Whilst the source of this material is unknown, it may be screened sands.

Suitable approaches for the restoration of each zone were identified (Table 2). For some zones, restoration would require primary and secondary works. A strategy for restoring the site was developed, based on 11 tasks (A–K) (see Table 3). In formulating the strategy we sought to make the most efficient use of vessels, to maximise use of materials available on-site, and to avoid unnecessary intervention.

Task A is the baseline survey, necessary to define the boundaries of physical and biological impact. Task B involves the removal of the sand waves in zone 4; this would be achieved by dredging, using a capital/maintenance style TSHD. The dredged material would then be placed within the trough of zone 2 (Task C), probably via the bottom doors whilst the vessel passed along the long-axis of the zone. The volume of sand within zone 4 is approximately the same as the volume of the trough. Task D involves capping the newly deposited sand within zone 2 with a layer of gravel. This would be achieved using the same TSHD, with suitable material for capping being sourced from a nearby licence area. Placement of this material could be achieved by either 'rainbowing' or 'jetting', a process in which material is effectively sprayed onto the target area. Task E involves an identical approach to cap the sandy sediments within zone 7. Tasks F, G and H involve bed levelling (using a TSHD) of zones 4, 1 and 3. Following a suitable period to allow for the natural dispersion of sands from zones 5/6, a survey of the site would then be undertaken to assess the success of restoration (Task I) and, importantly, whether it would be necessary to cap zones 5/6 (Task J). Task K is then an additional survey to assess success of restoration in zones 5/6.

3.2. Restoration costs

The overall costs of restoring Area 222 were assessed according to restoration works (per zone), licensing, carbon footprint and survey work and are presented in Table 4.

Table 1
Economic valuation techniques and examples of their relevance to ecosystem services.

Economic valuation method	Description	Relevance to ecosystem services
Choice Experiment Method (CEM)	Discrete choice model which assumes the respondent has perfect discrimination capability. Uses experiments to reveal factors that influence choice	Applicable to all ecosystem services
Contingent Valuation Method (CVM)	Construction of a hypothetical market by direct surveying of a sample of individuals and aggregation to encompass the relevant population. Problems of potential bias	Applicable to all ecosystem services
Cost-of-Illness (COI)	The benefits of pollution reduction are measured by estimating the possible savings in direct out-of-pocket expenses resulting from illness and opportunity costs	Applicable to: clean water and sediments; and immobilisation of pollutants
Damage Avoidance Costs (DAC)	The costs that would be incurred if the ecosystem good or service were not present	Applicable to: healthy climate; prevention of coastal erosion; sea defence; clean water and sediments; and immobilisation of pollutants
Defensive Expenditure Costs (DEC)	Costs incurred in mitigating the effects of reduced environmental quality. Represents a minimum value for the environmental function	Applicable to: healthy climate; prevention of coastal erosion; and sea defence
Hedonic Pricing (HP)	Derive an implicit price for an environmental good from analysis of goods for which markets exist and which incorporate particular environmental characteristics	Applicable to: tourism/nature watching
Market Analysis (MA)	Where market prices of outputs (and inputs) are available. Marginal productivity net of human effort/cost. Could approximate with market price of close substitute. May require shadow pricing where prices do not reflect social valuations	Applicable to: food; fish feed; ornamentals; medicine; aggregates; healthy climate; prevention of coastal erosion; and sea defence
Net Factor Income (NFI)	Estimates changes in producer surplus by subtracting the costs of other inputs in production from total revenue and ascribes the remaining surplus as the value of the environmental input	Applicable to: food, fish feed, medicines, aggregates, clean water and sediments; and immobilisation of pollutants
Production Function Analysis (PFA)	An ecosystem good or service treated as one input into the production of other goods: based on ecological linkages and market analysis	Applicable to: food; fish feed; ornamentals; medicine; aggregates; healthy climate; prevention of coastal erosion; and sea defence
Productivity Gains and Losses (PGL)	Change in net return from marketed goods: a form of (dose–response) market analysis	Applicable to: healthy climate; prevention of coastal erosion; and sea defence
Replacement/Substitution Costs (R/SC)	Potential expenditures incurred in replacing the function that is lost; for instance by the use of substitute facilities or 'shadow projects'	Applicable to all provisioning and regulating services but with limited role for cultural services
Restoration Costs (RC)	Costs of returning the degraded ecosystem to its original state. A total value approach; important ecological, temporal and cultural dimensions	Applicable to: healthy climate; prevention of coastal erosion; sea defence; clean water and sediments; and immobilisation of pollutants
Shadow Price of Carbon (SPC)	A price that reflects the social cost of carbon consistent with the damage experienced under an emissions scenario such that e.g. a specific policy goal can be achieved (the precautionary principle might support a further adjustment to the price)	Applicable to: healthy climate
Social Cost of Carbon (SCC)	Damage costs of an incremental unit of carbon (or equivalent amount of other greenhouse gas emissions) imposed over the whole of its time in the atmosphere	Applicable to: healthy climate
Travel Cost Method (TCM)	Cost incurred in reaching a recreation site as a proxy for the value of recreation. Expenses differ between sites (or for the same site over time) with different environmental attributes	Applicable to: tourism/nature watching

3.2.1. Licensing

The mass of deposits can be calculated from volumes using a conversion factor of 1.5 T m^{-3} for sand and 1.73 T m^{-3} for gravel (CE and BMAPA, 2010). The first campaign of restoration work involves three tasks where material will be placed on the seabed. These include Task C (placement of 135,019 m^3 of sand from zone 4 into zone 2), Task D (capping of zone 2 with 19,704 m^3 of gravel) and Task E (capping of zone 7 with 44,660 m^3 of gravel). The total mass of this material is 313,878 tonnes. This places the works in Band 4 (100,000–499,999 tonnes), incurring a licence fee of £19,850. This charge is divided between zones according to the mass of material deposited within each zone. The costs of deposition of sand within zone 2 have been associated with zone 4 as the only reason to infill this feature is to dispose of the sands within zone 4. As such it was appropriate that the portion of the licence fee should be applied to zone 4.

The second restoration campaign, if deemed necessary, would involve placement of 103,900 m^3/179,747 tonnes of gravel within zones 5/6. This again places the works in Band 4 (100,000–499,999 tonnes), incurring a licence fee of £19,850.

3.2.2. Carbon footprint

The Defra (2010b) methodology to calculate the traded cost of GHG emissions is used here under which the total quantity of emissions (as carbon dioxide equivalents, CO_2e) is calculated as the total quantity of fuel burnt (vessel time (days) multiplied by fuel consumption (tonnes per day) which for a TSHD is 4.7 tonnes/day (M. Russell, BMAPA, London, pers. comm.)) and a conversion factor for the quantity of GHG per tonne of fuel consumed (for gas oil, the fuel used by dredgers, the conversion factor is 3483.5 kg CO_2e per tonne). The total cost of emissions can then be calculated by multiplying the total quantity of CO_2e by the carbon price. For 2010, the 'traded price of carbon' per tonne was £14.10 (DECC, 2010). Table 4 shows the quantity and cost of GHG emissions (CO_2e) from the planned restoration works within each zone. Restoration of zones 5/6, if required, would increase the costs from £2,243 to £3,323. Gravel seeding of zones 5/6 would also account for the greatest quantity of emissions.

3.2.3. Survey

The costs for surveys are £54,834 for the baseline survey, £20,600 for the post-restoration survey, and £14,600 for the post-restoration survey of zones 5/6. Survey costs are equally distributed between zones, except for the post-restoration survey of zones 5/6.

3.2.4. Total costs

Assuming that sediments in zones 5/6 will naturally disperse then the total cost of restoration is £712,143 (Table 4). Most of this

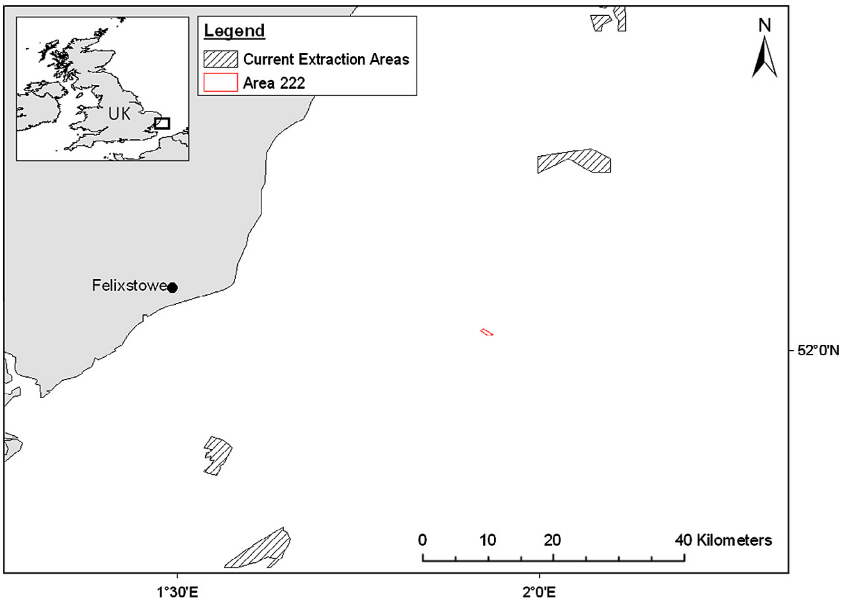

Fig. 3. Map of marine aggregate extraction areas in the Thames Estuary region.

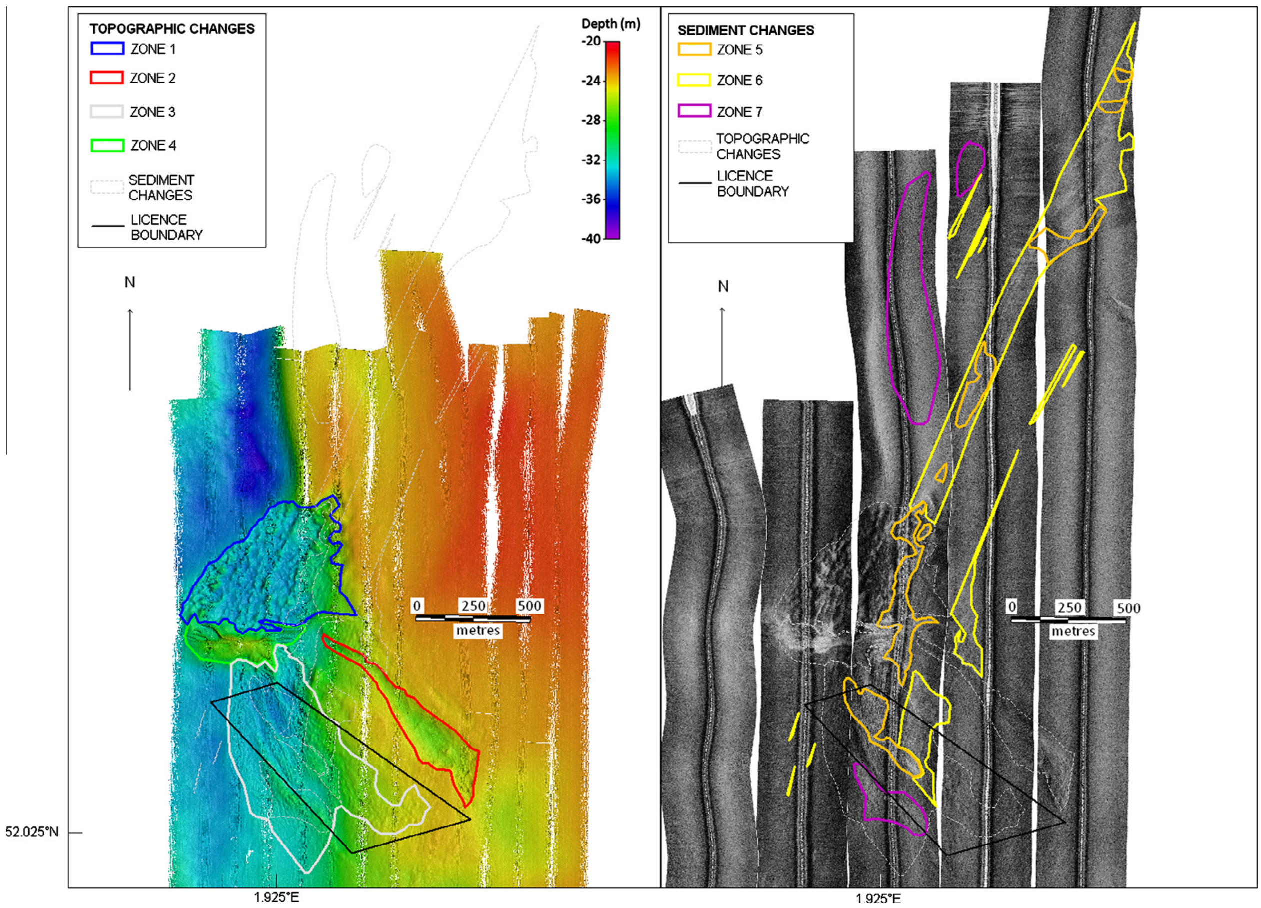

Fig. 4. (Part a) Topographic impact zones 1, 2, 3 and 4, overlaid on multibeam bathymetric data from 2004. Hashed lines indicate zones of sediment change (see part (b) for details). (Part b) Sediment change zones 5, 6 and 7, overlaid on sidescan sonar data from 2004. Hashed lines indicate zones of topographic change (see part (a) for details).

Table 2
Area, volume and suggested restoration approach(s) for each impact zone.

Impact	Zone	Description	Volume (m^3)	Area (m^2)	Restoration approach	
					Primary	Secondary
Topographic	1	Dredge depressions	−439,644	274,700	Bed levelling	–
	2	Trough	−135,019	98,520	Infill (material from zone 4)	Cap (gravel seeding)
	3	Dredge tracks	n/a	400,300	Bed levelling	–
	4	Sand waves	+135,541	68,640	Dredge (deposit in zone 2)	Bed levelling
		Total	−439,122	838,800		
Sediment	5	Rippled sand	n/a	159,338	Natural recovery	Cap (gravel seeding)?
	6	Sand ribbons/streaks	n/a	419,582	Natural recovery	Cap (gravel seeding)?
	7	Rippled sand	n/a	223,300	Cap (gravel seeding)	–
		Total	n/a	742,800		
		Grand total	−439,122	1,424,000		

Table 3
Restoration strategy – sequence of events.

Task	Action	Zone	Zone description
A	Baseline survey	All	
B	Dredging (deposit in zone 2)	4	Sand waves
C	Infill (material from zone 4)	2	Trough
D	Capping (gravel seeding)	2	Trough
E	Capping (gravel seeding)	7	Rippled sand
F	Bed levelling	4	Sand Waves
G	Bed levelling	1	Dredge depressions
H	Bed levelling	3	Dredge tracks
I	Post-restoration survey	All	
J[a]	Capping (gravel seeding)	5/6	Rippled sand and sand
K[a]	Post-restoration survey	5/6	

[a] Only where task I shows this to be warranted.

cost (86%) is attributable to the cost of restoration works, 10% to survey work, 3% for licensing and <1% with GHG emissions. This scenario will involve the relocation of 202,528 tonnes (135,019 m^3) of sand, and seeding of 111,350 tonnes (64,364 m^3) of gravel, equivalent to 1.09% of the 10.2 million tonnes extracted from the site.

If the restoration of zones 5/6 was deemed necessary then the total cost of works would rise to £1,189,660. In this case, 291,097 tonnes (168,264 m^3) of material would be required for seeding. This is equivalent to 2.85% of the total amount of material extracted from the licence.

The most expensive zone to restore in absolute terms would be 5/6, primarily due to its large size and the volume of capping material necessary. This zone, if restored, would account for 41% of the total cost. However, the most expensive zone, in terms of per square metre costs, is zone 4; at £3.49, this is more than 3.5 times more expensive than capping of zones 5/6.

3.3. Ecosystem services and goods/benefits

Table 5 qualitatively summarises the ecosystem services and goods/benefits provided by Area 222, and the likely impacts of aggregate extraction activities and potential impacts of restoration practices on these, based on expert opinion and literature (e.g. Austen et al., 2009). For several ecosystem services, extraction has had a negative impact on service provision while restoration may have a positive impact on provision (such as food and healthy climate). For other services our current knowledge is insufficient to assess the impacts of extraction and restoration activities on their provision (such as genetic resources and clean water and sediments). A summary of the available quantitative site-specific data, including valuation data where possible, is provided below for carbon sequestration, formation of species habitat, provision of fish and shellfish, food and aggregates, and the cultural service of education.

3.3.1. Carbon sequestration

Site specific biomass data can be used as an indicator of carbon sequestration, an intermediate ecosystem service provided by the marine environment (Fig. 2), albeit providing an underestimate given that it does not include mobile fish and shellfish species within the area. The use of biomass data as an indicator for carbon sequestration has been applied at other aggregate extraction sites (see Newell et al., 2004b; Austen et al., 2009), and allows an assessment of the impact of aggregate extraction on carbon sequestration. Data for Area 222 (using both 2010 traded and 2010 non-traded prices for carbon) show that the value of carbon at the low impact site ranges from £58.50 to £148.29 per km^2 over the time period whereas the prices at the high impact zones is much lower, ranging from £6.79 to £46.30 per km^2. In contrast, the shadow price of carbon at the non-impacted (reference) site ranged between £81.13 and £186.69 per km^2. The ranges are proportionately higher if the non-traded price is assumed (all values based on the 2010 traded price).

3.3.2. Formation of species habitat

Another intermediate service of importance to Area 222 is the formation of species habitat. The recent Outer Thames Regional Environmental Characterisation (REC, Emu Ltd., 2009) shows that gravelly sands and sandy gravels occupy 2419 km^2 and the impacted zones at Area 222 occupy 1.428 km^2 or 0.06% of this area. This habitat dominates the central area of the Outer Thames Estuary REC region and occupies 1382 km^2. The impact zones occupy 1.428 km^2 or 0.10% of this area. The impact zones at Area 222 also occupy circalittoral mixed sediment, a biotope complex which dominates the central area of the REC region and occupies 1345 km^2. The impact zones account for 1.428 km^2 or 0.12% of this area. The habitats present within the site may have a role as a spawning site for sole (*Solea solea*), and, to a lesser extent, thornback ray (*Raja clavata*), sandeels (*Ammodytidae* sp.) and plaice (*Pleuronectes platessa*) (Coull et al., 1998). It is uncertain whether the changes to the seabed at Area 222 will affect its capacity to act as a spawning ground for these species, although this is less likely for the planktonic spawners (e.g. sole and plaice) and more likely to affect those species where eggs are attached/buried within the substratum (e.g. sandeel and thornback ray). Thornback ray egg capsules are known to be deposited in sandy or muddy areas (Breder and Rosen, 1966). Hence changes may be positive or negative, depending on the species. The habitats present in Area 222 may also serve as a nursery ground for herring (*Clupea harengus*) and,

Table 4
Cost of restoration by zone, and for the site as a whole. The total cost for each zone is made up of separate costs for: restoration works, licensing, GHG emissions and survey work.

Zone	Description	Restoration Works			Licensing	Carbon Footprint		Survey			Total cost	% of total cost (Inc. restoration zones 5/6)	Cost/m^2
		Dredging	Capping	Bed levelling		CO_2e (tones)	Cost 'traded' price	Baseline	1st Post-restoration survey	2nd Post-restoration survey			
1	Dredge depressions	–	–	£77,000	–	25.21	£356	£9,139	£3,433.3	–	£89,928	13 (8)	£0.33
2	Trough	–	£64,855	–	£2,156	8.36	£118	£9,139	£3,433.3	–	£79,701	11 (7)	£0.81
3	Dredge tracks	–	–	£112,500	–	36.84	£519	£9,139	£3,433.3	–	£125,591	18 (11)	£0.31
4	Sand waves	£193,763	–	£19,500	£12,808	69.82	£984	£9,139	£3,433.3	–	£239,627	34 (20)	£3.49
5/6	Rippled sand & Sand ribbons/streaks	–	£441,987	–	£19,850	76.60	£1,080	£9,139	£3,433.3	£14,600	£12,572–£490,089	2 (41)	£0.02–£0.94
7	Rippled sand	–	£146,998	–	£4,886	18.86	£266	£9,139	£3,433.3	–	£164,722	23 (14)	£0.74
	Total (ex zones 5/6)	£193,763	£211,853	£209,000	£19,850	159.1	£2,243	£54,834	£20,600	–	£712,143		
	Total (Inc. zones 5/6)	£193,763	£653,840	£209,000	£39,700	235.7	£3,323	£54,834	£20,600	£14,600	£1,189,660		

to a lesser extent, cod (*Gadus morhua*), whiting (*Merlangius merlangus*), mackerel (*Scomber scombrus*), sole, plaice, tope (*Galeorhinus galeus*) and sandeels. The nature of the habitat provided by each of the zones will determine their suitability as nursery areas for individual species.

3.3.3. Fish and shellfish

Area 222 is likely to provide fish and shellfish. Probability distribution maps for key commercial species (Stelzenmüller et al., 2010) indicate the likely target fish communities present within Area 222 and the wider Outer Thames Estuary REC (see Fig. 5) include the flatfish sole, thornback ray and plaice; the gadoids cod and whiting, and the bivalve mollusc queen scallop (*Aequipecten opercularis*). These are associated with particular habitat types and their presence provides final services in the form of fish and shellfish stocks. As there are no data on the commercial fish populations within the Area 222 physical impact zone, we can only qualitatively indicate the species present. We acknowledge that quantitative evidence, together with errors, would provide a stronger basis for monetary valuation of the commercial species present within the site. In addition, recent evidence suggests that entrainment of benthic fish species as a result of dredging activities may also have a significant impact on local fish populations (Drabble, 2012) although site-specific evidence was not available for Area 222.

3.3.4. Food

One of the goods/benefits provided by Area 222 relates to the provision of food obtained from commercial fisheries. The most recent assessment of the location and intensity of inshore and offshore fishing activity in the Thames REC region is provided by Vanstaen et al. (2010). This study integrates inshore maps of fishing activity, based on Kent and Essex Inshore Fisheries and Conservation Authority (formerly the Kent and Essex Sea Fishery Committee) observation data, and maps of offshore fishing activity, based on an assessment of Vessel Monitoring System (VMS) data. The resulting map of fishing activity, based on data from 2007 to 2008, suggests the only significant fishing activity from UK vessels in the vicinity of the study site is associated with beam trawling and otter trawling by vessels over 15 m in length. This allows us to determine whether there may be any avoidance, or even targeting, of the impacted areas. Most of the fishing activity is to the west of the site, and clearly there is some overlap of activities. Whilst there does not appear to be much fishing activity across the impacted areas, there is some evidence of otter trawling within zones 1, 2, 3, 5 and 6. In contrast, only one incidence of beam trawling was identified on the western margins of zone 1. Without pre-dredging data it is not possible to determine whether fishing patterns have changed in response to the impacts associated with dredging at Area 222.

An indication of the economic value of fishing activities is available from the Marine Management Organisation (MMO) in the form of landing values. However, as these data are recorded at the scale of ICES rectangles, they are of little relevance for the purposes of the current study although by combining landings and VMS data, it is possible to approximate the value of fish at a much finer resolution (see ABPmer, 2009). The annual mean value of trawled fish within the Thames REC region was calculated to be £186,273 which suggests the annual value of fish within Area 222 might be considered negligible.

3.3.5. Aggregates

One of the key goods/benefits provided by Area 222 is the raw materials from aggregate extraction. During the term of the licence 10.2 million tonnes of aggregate were extracted from Area 222. Although annual landing prices between 1972 and 1996 are not available, based on a present day average price of aggregate of £12 per tonne (2010 prices ex wharf) (M. Russell, BMAPA, London, pers. comm.), this equates, at current prices, to over £120 million of aggregates being extracted from the site during the licence term. It has also been recognised that society gains additional benefits from the extraction of these raw materials in employment associated with the operating and servicing of dredging vessels, in wharves, offices and administration and in the delivery, transport and use of the aggregates (Highley et al., 2007; Austen et al., 2009). Information on the use of the aggregates extracted from Area 222 is not available and therefore such benefits cannot be accurately quantified. However, attributing specific benefits associated with its actual use may be misleading as alternative sources of aggregates are typically available, albeit at a higher private or social cost, for example from land-based sources. Similarly, the values will vary widely depending on how the aggregates are used. For example, if they are used for beach nourishment or building there are further economic benefits.

Table 5
Identification of ecosystem services and goods/benefits provided by Area 222, including an indicative assessment of the impact of aggregate extraction and restoration assuming the state of the site was post-extraction with no prior attempt at restoration.

Intermediate services	*Impacts of extraction/ restoration	Final services	*Impacts of extraction/ restoration	Goods/benefits	*Impacts of extraction/ restoration
Supporting services		*Provisioning services*		*Provisioning*	
Primary production	−/+	Fish/shellfish	−/+	Food	−/+
Laval/gamete supply	−/+	Other wild food	0/0	Fish feed	?/?
Nutrient cycling	?/?	Seaweed (fertiliser)	0/0	Ornamentals	?/?
Water cycling	?/?	Ornamental material	?/?	Medicine	?/?
Formation of species habitat	−/+	Genetic resources	?/?	Aggregates	−/0
Formation of physical barriers	0/0				
Formation of pleasant scenery	−/+				
Regulating services		*Regulating services*		*Regulating*	
Biological control	?/?	Climate regulation	−/+	Healthy climate	−/+
Natural hazard regulation	0/0	Natural hazard protection	0/0	Prevention of coastal erosion	0/0
Regulation of sediment and water quality	−/+	Waste breakdown	−/+	Sea defence	0/0
Carbon sequestration				Clean water and sediments	?/?
	−/+			Immobilisation of pollutants	−/?
		Cultural services		*Cultural*	
		Meaningful places	−/+	Tourism/nature watching	0/0
		Socially valued seascapes	−/+	Spiritual/cultural wellbeing	−/+
				Aesthetic benefits	−/+
				Education	0/0

* KEY: Impact of extraction/Impact of restoration; + = positive impact; − = negative impact; 0 = no/negligible impact; ? = unknown impact.

3.3.6. Education

Another class of good/benefit considered to be of importance with respect to Area 222 is education, relating to the marine environment. It is estimated that £187,500 of funded research has taken place (between 2000 and 2012) specifically referring to the impact of dredging at Area 222 and such studies have included several research papers (Boyd et al., 2003, 2004, 2005; Wan Hussin et al., 2012).

4. Discussion

This study aimed to estimate the cost of physical restoration at the scale of a former extraction site (Area 222), to assess whether such action could be justified and also to show the feasibility of such an economic valuation exercise. The acoustic data collected in 2004, 8 years after cessation of dredging (Cooper et al., 2005), show seven zones of persistent physical impact at the study site characterised by either changes in topography, sediment composition, or a combination of both. Hence a restoration plan was developed for the site to address the impacts observed within each zone. The aim of restoration was to create a more level seabed, and to restore the sediment character. Three restorative techniques were identified as being potentially suitable for this purpose: dredging/disposal, bed levelling, and capping. Dredging/disposal was deemed a suitable technique to move large volumes of material from higher to lower areas; bed levelling using a TSHD, for restoring small-scale topographic undulations, and capping, using a gravel-seeding approach (Cooper et al., 2011c), for restoring sediment character. The estimated predicted total cost of restoring the site was £712 k–£1,189 k, depending on whether natural recovery occurred following the removal of a sand wave feature in the west of the site.

In seeking to understand whether this sum could be justified we assessed the significance of the persistent impacts on the ecosystem services and goods/benefits offered by the site. The methodology, which incorporates the DPSIR framework and encompasses The Ecosystem Approach, proved useful in identifying relevant issues. Full economic valuation (such as recommended by Turner et al. (2003)) of the goods/benefits was not possible due to the lack of valuation data, and difficulties in establishing the appropriate boundaries for assessment, a constant difficulty in open marine areas. However, the available data showed that several ecosystem services and goods/benefits have been affected, and it is predicted that gains may be achieved following restoration. It is important to note the potentially conflicting outcomes associated with restoration in relation to spawning and fisheries issues. For example, the creation of bathymetric depressions was thought to present an opportunity for gadoid fish, through the provision of shelter (Alex Simpson, Cefas fishing skipper, pers. comm.). It is recommended that potentially positive outcomes such as this are considered, and even planned in at the EIA stage. In the Netherlands this approach is referred to as ‘Building with Nature’ (De Vriend and Van Koningsveld, 2012).

On balance, our analysis indicates that the restoration of the seabed at the study site would not be justified. This site-specific decision was based not only on the consideration of ecosystem services and goods/benefits but also on the absence of a licence condition concerning the status of the seabed after dredging, and the lack of pre-dredge data.

This study has been important for a number of reasons. Firstly, the findings and approach taken could be used at other sites in order to identify, objectively, whether it is appropriate to attempt to restore the seabed where impacts persist, what techniques should be used and how success can be judged. Secondly, it indicates that a generic restoration plan can be developed. Here we primarily consider physical restoration. Whilst acoustic data proved excellent for identifying physical impacts, we suggest impacted zones should also be ground-truthed for biology and sediment characterisation; for example, Boyd et al. (2005) importantly showed faunal recovery within an area of the site where some physical impacts of dredging remained which suggests that a complete physical recovery is not necessarily a pre-requisite for faunal recovery. Cooper (2012) further describes a method for identifying where changes in sediment composition are likely to have an ecological significance in terms of the capacity for faunal recovery post-dredging. Had suitable ground-truth data been available for this study, it is likely that the size of some of the zones identified as impacted may have been smaller, reducing the overall cost of restoration. This highlights the need to have a clear goal when restoring, whether it is to restore the physical environment, the biology or both (Elliott et al., 2007). In terms of restoring sediment composition, Cooper (2012) shows it is not necessary to recreate exactly the pre-dredge sediment conditions, merely to return them to within the range naturally found in association with the pre-dredge faunal assemblage in the wider region.

Thirdly, we have identified several potential techniques for physically restoring a site although there is still a poor understanding of whether such techniques would be successful. Future studies should consider the practicality and effectiveness of bed levelling using both a TSHD and dredge plough, and capping using a process of reverse screening to return a layer of coarse sediment to the seabed surface after dredging.

Fourthly, the study indicates cost of restoration for a former extraction site. Whilst the costs are clearly large, they need to be considered in relation to the value of material extracted. Unfortunately we were unable to make any assessment as to the affordability of restoration for a developer, as relevant figures are not in the public domain. However, given that the value of the material extracted was estimated as £122.4 million, then restoration costs may equate to between 0.58% and 0.97% of that value. Such estimates are valuable in discussions concerning affordability. Given such likely costs, it is important to consider how restoration works could be funded. One option would be to follow the approach taken on land, where costs are borne by the developer (Bellew and Drabble, 2004). Another potential source of funds would have been the now discontinued Aggregate Levy Sustainability Fund (ALSF), which aimed to reduce the environmental impacts of the extraction of aggregates and to deliver benefits to areas subject to these impacts (Defra, 2010a).

Fifthly, use of the DPSIR framework together with an ecosystem services approach has shown to be valuable in identifying potential impacts on the wider ecosystem. However, the lack of valuation data means that the approach is, at present, unlikely to provide the information required for a full CBA to help determine whether restoration is justified. As an alternative, it is important that scientifically justifiable and measurable licence conditions for acceptable change in sediment composition following marine aggregate dredging are imposed (see Cooper, 2012, 2013).

Lastly, evidence from the study site suggests it may be difficult to require developers to restore sites of historic dredging activity especially given the absence of pre-dredge data; sites licensed before the adoption of the original European EIA Directive in 1985 are unlikely to have had any associated environmental monitoring. Hence there is uncertainty about whether all features identified post-dredging are dredging related. In addition, many older

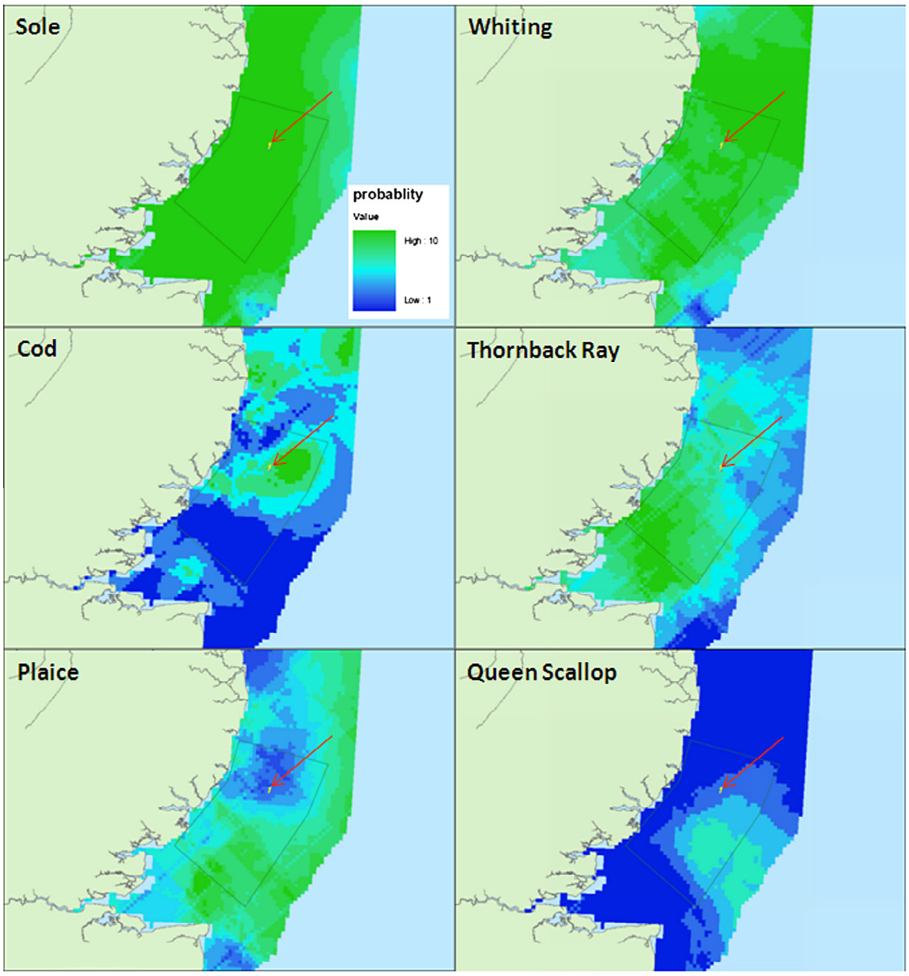

Fig. 5. Probability distribution maps for key commercial species where probability values is greater than 1 (adapted from Stelzenmüller et al., 2010). The red arrow indicates the position of the topographic and sediment change impact zones from the present study.

licences do not have licence conditions governing the state of the seabed post-dredging. As a result, restoration of historic sites would require funding, possibly from a levy system. However, any restoration of historic sites could undo the natural recovery which has taken place especially as the natural and unaided recovery of open marine sites is likely given time (Elliott et al., 2007).

These issues are likely to be less relevant to recent licences where regular environmental monitoring allows management actions to prevent unacceptable impacts from developing, and where a licence condition requires developers to leave the seabed in a 'similar' physical condition after dredging. However, this licence condition is potentially problematic as it relies on a subjective assessment of the similarity of the seabed before and after dredging. Cooper et al. (2011a) and Elliott (2011) identified that such licence conditions and monitoring need to be specific and measurable, so it is clear when the condition has not been met, and appropriate management actions can be taken.

In addition to the need to further investigate the practicality and effectiveness of the restoration techniques identified in this study, further work is also required to increase our understanding of recovery times for extraction sites in a range of environmental conditions. Borja et al. (2010) indicate the range of recovery times for differing stressors and biological components and they and Elliott et al. (2007) suggest that these are reduced for open, interconnected systems where the physical character is recovered and the biota have mobile recruitment stages. Hence it is important to identify sites where impacts are likely to persist, and hence where restoration might be considered, and sites where impacts are likely to recover naturally. Such information will improve the quality of EIAs and decision making. In addition, the consideration of ecosystem services and goods/benefits highlights a need to consider the wider significance of localised changes to the seabed resulting from marine aggregate dredging. Recent work has begun to address these questions (e.g. Kenny et al., 2010; Daskalov et al., 2011), but this is an area where further work is required. In addition, we need more valuation data (e.g. ABPmer, 2009) to enable better decision-making using a cost-benefit approach (Defra, 2007).

5. Conclusion

Given the lack of understanding concerning the significance of localised dredging-related impacts for the wider ecosystem, under the precautionary principle it is necessary to consider restoring them when they occur or more importantly preventing those impacts, for example by not over-exploiting an area such that the colonising fauna do not have a suitable substratum to occupy. The current system of ongoing monitoring during the life of the licence allows the status of the seabed to be reviewed at regular intervals and action to be taken where necessary. There is, however, a need to define better the licence conditions for acceptable seabed condition, and to have a system of feedback monitoring (Gray and Elliott, 2009) which allows for early management intervention should unacceptable seabed conditions develop (see Cooper, 2013). For older licenses, where there was neither a requirement for environmental monitoring nor a licence condition relating to the status of the seabed post-extraction, it is difficult to see how individual developers could be required to undertake restoration. For these areas, a need for restoration, were it ever deemed necessary, is most likely to be identified at a regional level, possibly in response to future marine spatial planning and ecosystem management.

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Paper #3

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Setting limits for acceptable change in sediment particle size composition following marine aggregate dredging

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ABSTRACT

In the UK, Government policy requires marine aggregate extraction companies to leave the seabed in a similar physical condition after the cessation of dredging. This measure is intended to promote recovery, and the return of a similar faunal community to that which existed before dredging. Whilst the policy is sensible, and in line with the principles of sustainable development, the use of the word 'similar' is open to interpretation. There is, therefore, a need to set quantifiable limits for acceptable change in sediment composition. Using a case study site, it is shown how such limits could be defined by the range of sediment particle size composition naturally found in association with the faunal assemblages in the wider region. Whilst the approach offers a number of advantages over the present system, further testing would be required before it could be recommended for use in the regulatory context.

1. Introduction

In the United Kingdom, companies extracting marine aggregate (sand and gravel) from the seabed are typically required, through a condition attached to the extraction licence, to leave the seabed in a similar physical condition after the cessation of dredging. This requirement, articulated in the government policy document covering marine aggregate dredging (ODPM, 2002), is intended to promote recovery, and the return of a similar faunal community to that which was present before dredging, thus reducing the likelihood of long-term and potentially cumulative impacts on the wider ecosystem.

Evidence suggests this policy is sensible. For example, numerous studies (Desprez, 2000; Newell et al., 2004a,b; Boyd et al., 2005; Robinson et al., 2005; Desprez et al., 2010; Cooper et al., 2011a,b; Barrio Frojan et al., 2011; Wan Hussin et al., 2012) have shown that a change in sediment composition, typically a result of sustained screening, where unwanted sediment fractions are returned to the seabed, has the potential to alter the benthic community composition, and the potential for recovery. In such cases it is difficult to argue that seabed sediments have remained in a 'similar' condition post dredging. Whilst this policy is clearly sensible, and in line with the principles of sustainable development, the ambiguity associated with the term 'similar' can be problematic for both developer and regulator. For example, developers have no clear definition of what is acceptable or not in terms of changes in sediment composition, and the regulator is forced to make subjective assessments as to the acceptability of changes that may occur. If the condition is to achieve its purpose of mitigating adverse environmental impacts then it has to be enforceable, and this requires it to be specific and measurable. The development of a more objective method of assessment is timely given that approximately forty licence applications for marine aggregate extraction are expected in the next 2–3 years; this is a result of many existing licences reaching the end of their previous terms, and a new licensing system resulting from the Marine and Coastal Access Act (2009).

A possible solution to this problem comes from recent Regional Environmental Characterisation (REC) initiatives which have mapped the biological resources present within, and surrounding areas of marine aggregate dredging. The first of these surveys was undertaken in 2002 in the Eastern English Channel (James et al., 2007). Subsequent surveys have occurred off the South Coast (James et al., 2010), Thames (Emu Ltd., 2009), East Coast (Limpenny et al., 2011) and Humber regions (Tappin et al., 2011). These surveys, and more localised habitat mapping initiatives (e.g. Boyd et al., 2004; Brown et al., 2004; Birchenough et al., 2010; ECA and Emu Ltd., 2010; ERM, 2010), have provided a new understanding of the distribution of benthic faunal communities in regions of marine aggregate dredging. In addition, and crucially, the collected data make it possible to identify the range of sediment conditions found in association with individual benthic faunal assemblages typical of marine aggregate producing regions; it is this information which has the potential to define limits for acceptable environmental change. In theory, as long as the composition of sediment within an impacted area remains within an acceptable range, as defined by the initial pre-dredge state and comparable

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conditions in the wider region, then a return of an acceptable benthic assemblage should be possible following the cessation of dredging. Such an approach fits well with results reported by Cooper et al. (2011a), which showed that the sensitivity of benthic faunal assemblages to changes in sediment composition caused by marine aggregate dredging is site specific. The suggested approach would allow for this, providing an appropriate level of localised protection. The acceptable change limits would be identified by the regulator and set out in a proposed licence condition during the Environmental Impact Assessment phase of the development. Having established the condition for acceptable change in sediment composition, this would become a focus for the developer lead monitoring, and final post-dredge assessment of seabed status.

The aim of this study was to evaluate the approach, using data from an existing broad scale survey undertaken in the region of the Hastings Shingle Bank extraction area, located off the south coast of the UK (Boyd et al., 2004). The objectives were to: (i) identify, characterise and map the broad scale distribution of the faunal assemblages present in the survey area, (ii) identify the range of sediment particle size composition found in association with each faunal assemblage, (iii) identify the faunal assemblage(s) likely to have been present before dredging within the extraction area, (iv) identify a suitable licence condition for acceptable change in sediment composition within the dredged area, (v) assess compliance with the stated condition at the time of the survey.

2. Methods

The data used in this study were obtained in 2003 for the purpose of producing a biotope map of the Hastings Shingle Bank (see Boyd et al., 2004). Macrofaunal and sediment samples were acquired, using a 0.1 m^2 Hamon grab, from a total of thirty-four stations within the limits of the side scan sonar survey (see Fig. 1), and samples were processed in accordance with guidelines set out in Department for Transport, Local Government and the Regions (2002). Of the thirty-four stations sampled, twenty-six were located outside the licensed extraction area. The remaining eight stations were located inside the extraction area (see Fig. 1).

2.1. Faunal assemblage distribution

2.1.1. Approach

The biotope map of the Hastings Shingle Bank found in Boyd et al. (2004) was considered unsuitable for the purposes of this study as it shows the benthic faunal assemblages found in association with different physical habitats (i.e. habitat was the starting point for the biological map). For the purposes of the present study, the interest is in identifying what physical conditions (i.e. sediment composition) are found in association with a specific faunal assemblage type (i.e. the faunal assemblages are the starting point); it was therefore necessary to undertake a reanalysis of the data.

2.1.2. Analysis

Multivariate community analyses were carried out using the Primer 6® package (Clarke and Warwick, 1994). Macrofaunal abundance data (i.e. excluding colonial species) from the twenty-six samples taken outside the licence area were subjected to a square-root transformation to down-weight the dominance of very abundant taxa, thereby giving rarer taxa a greater influence in the subsequent analyses (Clarke and Green, 1988). Similarities were then calculated between all sample pairs using the Bray–Curtis coefficient of similarity (Bray and Curtis, 1957), prior to group average clustering of the data. Following clustering, a series of ‘similarity profile’ (SIMPROF) permutation tests were used to look for statistically significant sample clusters in the data. Samples

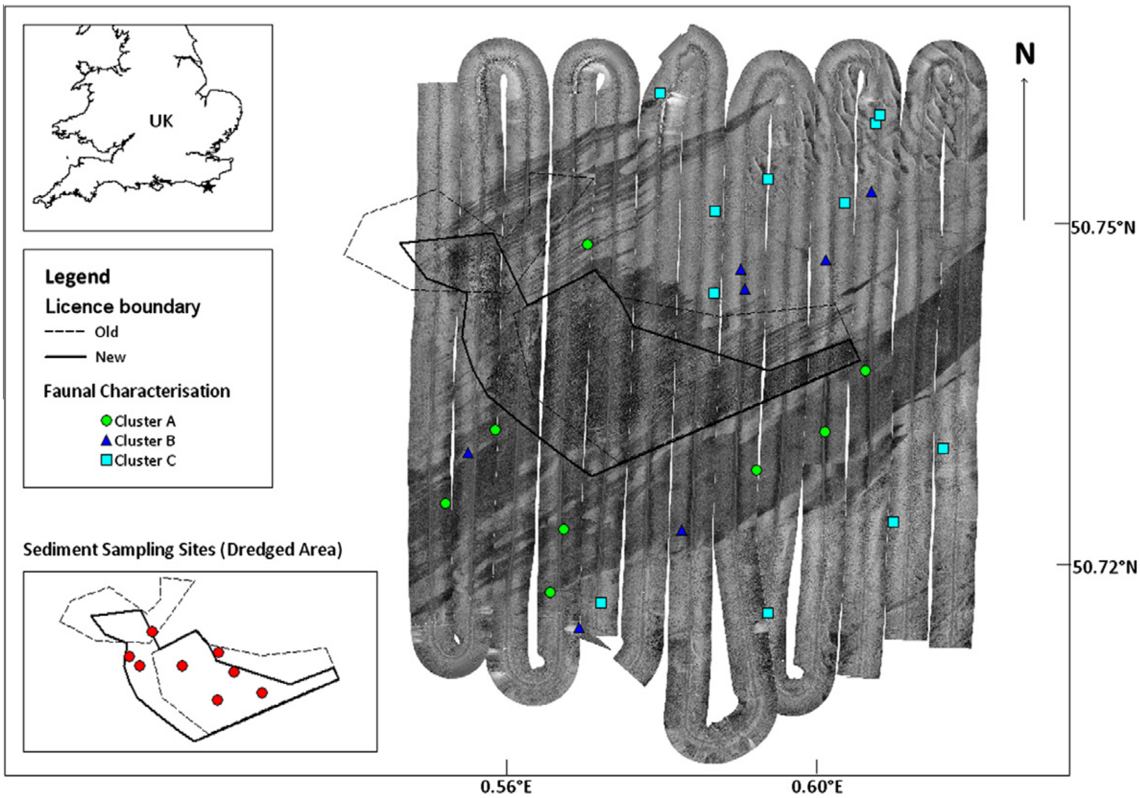

Fig. 1. Distribution of the three faunal assemblages (cluster groups A–C) overlaid on the sidescan sonar data. Inset at bottom left shows the location of sediment sampling stations within the dredged area.

taken from within the licence boundary were excluded from the analysis, as the faunal assemblage here was likely to be modified by ongoing dredging; clearly this would not be necessary in a new dredge application area. In a Geographic Information System (GIS), sampling stations, identified by their cluster group, were overlaid on the acoustic data and licence boundary. This allowed the distribution of faunal communities, and their association with different habitats, to be seen.

2.2. Cluster group characteristics

2.2.1. Fauna

The number of species (including colonial taxa) and individuals (excluding colonial taxa) were determined for each of the twenty-six samples taken from outside the dredging area. A scatter graph was used to examine the difference in both metrics between the different faunal cluster groups. Characterising species for each group were identified using the SIMPER routine in Primer.

2.2.2. Sediment

Plots of sediment particle size distribution were used to compare the composition of samples belonging to each of the identified faunal cluster groups. Plots consisted of the cumulative percentage weight by 0.5 phi (ϕ) sediment intervals; the phi scale is a logarithmic measure of sediment grain size, ranging from -8 (>256 mm) to >8 (1/256 mm). For each cluster group, the mean and 95% confidence intervals of the cumulative distribution were also plotted due to the small number of sample replicates. Confidence intervals were calculated using a rearrangement of the equation used to calculate standard or *z*-scores:

$$x = \mu \pm (z * \sigma)$$

Where μ is the mean value, x is the individual raw score (based on the upper or lower limit of the 95% confidence interval), σ is the standard deviation, and z is the 97.5 percentile point of a normal distribution (=1.96).

Using the cumulative sediment data, the percentage contribution of major sediment fractions (% gravel, % coarse sand, % medium sand, % fine sand, % silt/clay) were calculated for individual samples. For each of the above sediment classes, a Kolmogorov–Smirnov test was used to test for normality of the data. Using these summary data, the mean $\bar{x}$, and the upper and lower limits of the 95% confidence intervals ($\bar{x}$ critical) were again calculated; based on the normal distribution, the true population mean would be expected to fall within the upper and lower $\bar{x}$ critical values. The confidence intervals therefore define the range of sediment composition, in terms of the major sediment classes, found in association with each faunal cluster group.

In order to assess whether dredged samples fell within the acceptable range, it would be necessary to have sufficient statistical power ($1 - \beta$) (Green, 1989) to detect a change equivalent to $\bar{x}$ critical minus $\bar{x}$. Power analysis, for a two-sample *t*-test, was used to determine the statistical power available based on the eight samples belonging to cluster group A. This analysis was undertaken in Minitab™ v15, using input variables for standard deviation (s), sample size (n), and the difference to be detected (i.e. $\bar{x}$) critical minus $\bar{x}$). Where the required level of statistical power is entered (and the sample size field left blank), the analysis tool returns the required number of samples.

An ANOSIM test was used to determine whether there were statistically significant differences between the sediment compositions of samples belonging to the three cluster groups; the data used in this analysis were untransformed sediment weights by 0.5 phi interval. The R value from this test provides a measure of the difference between groups, and would be expected to be in the range from zero to one; a value of zero implies there is no difference between groups, whilst a value of one implies that groups are completely different. An associated *p*-value of <0.05 was taken to imply statistical significance.

2.3. Setting the licence condition

The identity and spatial distribution of different faunal assemblages within the area of influence of dredging would normally, for a new licence area, be determined before the onset of dredging. However, as the data used in this study were taken from an active dredge site, this was not possible. We therefore identified the most likely faunal assemblage to have been present with the dredged area before dredging based on the distribution of sediments, bedforms and faunal assemblages in the wider region. This approach to retrospectively identify the probable pre-dredge faunal communities will be highly relevant to licence renewals, where no pre-dredge data exists. As the site has not be subject to screening, an assumption was made that the footprint of effect was confined to the licensed area itself; the broad scale side scan sonar and map of faunal assemblage distribution provide evidence to support this assumption.

Based on the faunal assemblage(s) thought to have been present within the licence area, a proposed licence condition was written, requiring the composition of sediments, in terms of the major sediment fractions (% gravel, % sand, % silt/clay), to remain within the range, as defined by the 95% confidence limits, of sediment composition found in association with the relevant assemblage in the wider region.

2.4. Assessing compliance with licence condition

The location of sample stations within the dredged area is shown in Fig. 1. Sediment samples taken from these stations were used to assess whether the composition fell within the range seen in the wider region for the most likely pre-dredge faunal assemblage(s). A comparison of the sediment distribution data (including mean and 95% confidence intervals) for dredged, and cluster group A samples was used to provide a visual assessment of whether sediments appeared to be the same.

An objective assessment of compliance with the licence condition was achieved by comparing the means (% gravel (2.0–45.0 mm), % sand (0.063–1.4 mm) and % silt/clay (0.00010–0.0442 mm) of samples belonging to the dredged, and cluster group A treatment groups. This analysis was performed using a 'Student's' two-sample *t*-test with the following hypotheses:H_0: $\bar{x}_{(CLUSTERA)} = \bar{x}_{(DREDGED)}$$H_1$: $\bar{x}_{(CLUSTERA)} \neq \bar{x}_{(DREDGED)}$Under the Null hypothesis (H_0), the means of the two sample groups are the same, whereas under the Alternative hypothesis (H_1), the means of the two sample groups are different. As the difference could be in either direction a two-tailed test was applied. Where the test showed no difference between the groups (i.e. $p > 0.05$) then H_0 was accepted. In this case, sediments within the dredged site were assessed to have met the condition for acceptable change. Where $p \leqslant 0.05$ then H_0 was rejected in favour of H_1, representing an unacceptable change. *T*-tests were carried out using Microsoft excel.

Whilst the power analysis carried out using the cluster A data indicates that the eight sediment samples should provide sufficient statistical power to detect the necessary level of change for each sediment fraction (i.e. $\bar{x}$ critical minus $\bar{x}$), an additional power analysis was undertaken using the pooled standard deviation, based on the sediment samples from both groups.

Finally, an ANOSIM test, based on untransformed particle size data (% sediment weight by 0.5 phi interval), was used to confirm the results of the *t*-tests, and to determine whether dredged sediments, like those from cluster group A, remained distinct from those belonging to the other cluster groups observed within the region. As data were all in the same units, it was not necessary to undertake a normalisation of the data prior to undertaking this analysis.

3. Results

3.1. Faunal assemblage distribution

The result of cluster analysis performed on the macrofaunal data is shown in Fig. 2, with the SIMPROF test revealing 4 statistically distinct faunal groups at $p \leqslant 0.05$. On the basis of the dendrogram, the single sample within the first cluster was added to cluster group A. This left three cluster groups (A–C), the distribution of which are shown, overlaid on the side scan sonar data, in Fig. 1. Based on the underlying side scan data, faunal group A was found in association with coarser sediment, as indicated by a darker area, whereas faunal groups B and C were found in areas of finer sediment, as shown by lighter patches on the acoustic record.

3.2. Cluster group characteristics

3.2.1. Fauna

Comparison of the univariate summary measures reveals higher numbers of species and individuals in faunal cluster group A compared with groups B and C (Fig. 3). The results of the SIMPER test (Table 1) revealed that the richest assemblage – faunal cluster group A – was characterised by species typically associated with gravel dominated sediments (e.g. the crustaceans *Upogebia* sp., *Balanus crenatus*; polychaetes *Lumbrineris gracilis* and *Pomatoceros lamarcki*). In contrast, characterising species of faunal cluster groups B and C were more typical of mobile sandy sediments. Characterising species in faunal cluster group B included the polychaete *Nephtys* sp., and a high proportion of crustaceans (e.g. the amphipod *Bathyporeia elegans*, and the opossum shrimp *Gastrosaccus spinifer*). In contrast, characterising species in faunal cluster group C included a higher proportion of polychaetes (e.g. *Notomastus* sp., *Nephtys* sp. and *Ophelia borealis*), and the echinoderm *Echinocyamus pusillus*.

3.2.2. Sediment

Plots of sediment distribution clearly show that the different faunal cluster groups are associated with different sediments types (Fig. 4). Sediments associated with faunal cluster group A were dominated by gravels, whilst groups B and C were dominated by sands. Group B differed from group C in having a slightly higher proportion of coarse sediments.

The results of an ANOSIM test (Table 2a) showed that the apparent differences between cluster groups A–C evident in Fig. 4 were statistically significant ($p < 0.05$). From this we deduce that, by implication, any significant long-term alteration in sediment composition within the dredged area would be likely to result in a change in the faunal composition.

3.3. Setting the licence condition

Based on the existence of faunal cluster group A within undisturbed gravel deposits surrounding the licence area, we suggest that this group was most likely to have been present within the dredged area before the onset of operations. The range of sediment composition found in association with assemblage A, as defined by the 95% confidence intervals, was therefore to be used to assess whether sediment changes within the dredged areas fell within acceptance limits.

The results of a power analysis showed that, based on the eight samples collected from cluster group A, there was sufficient statistical power ($1 - \beta$) to assess differences at the level of % gravel, % sand and % silt/clay. Assessing differences at the finer resolution (% gravel, % coarse sand (0.5–1.4 mm), % medium sand (0.250–0.355 mm), % fine sand (0.063–0.180 mm), % silt/clay) was considered inappropriate as there was insufficient statistical power to assess the level of change associated with the medium sand fraction (see Table 3, Power 1).

Given the above results, the licence condition for the dredged areas in the present study would state:

'Within the confines of the dredged area on the Hastings Shingle Bank, the mean composition of sediments must, after cessation of dredging, and allowing a suitable time for recovery, be left in the range: Gravel 38–73%, Sand 25–55%, Silt/Clay 2–7%.

3.4. Assessing compliance with the licence condition

Whilst the mean sediment profile of the dredged sediments was similar to that of the mean sediment profile for samples belonging to cluster group A, there was some evidence of a higher variability within the dredged area (Fig. 5).

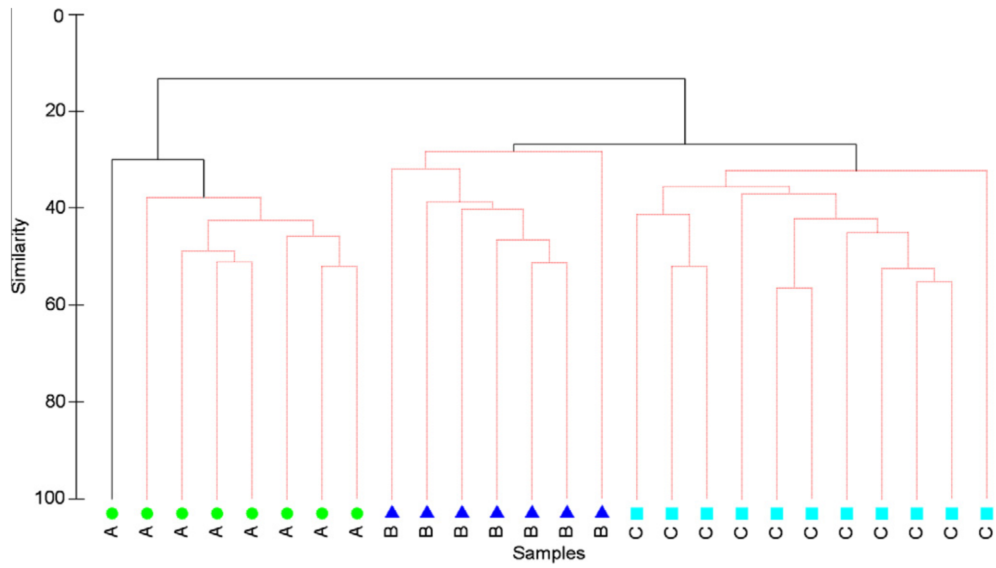

Fig. 2. Dendrogram resulting from a cluster analysis of square-root transformed macrofaunal abundance data (colonial taxa excluded).

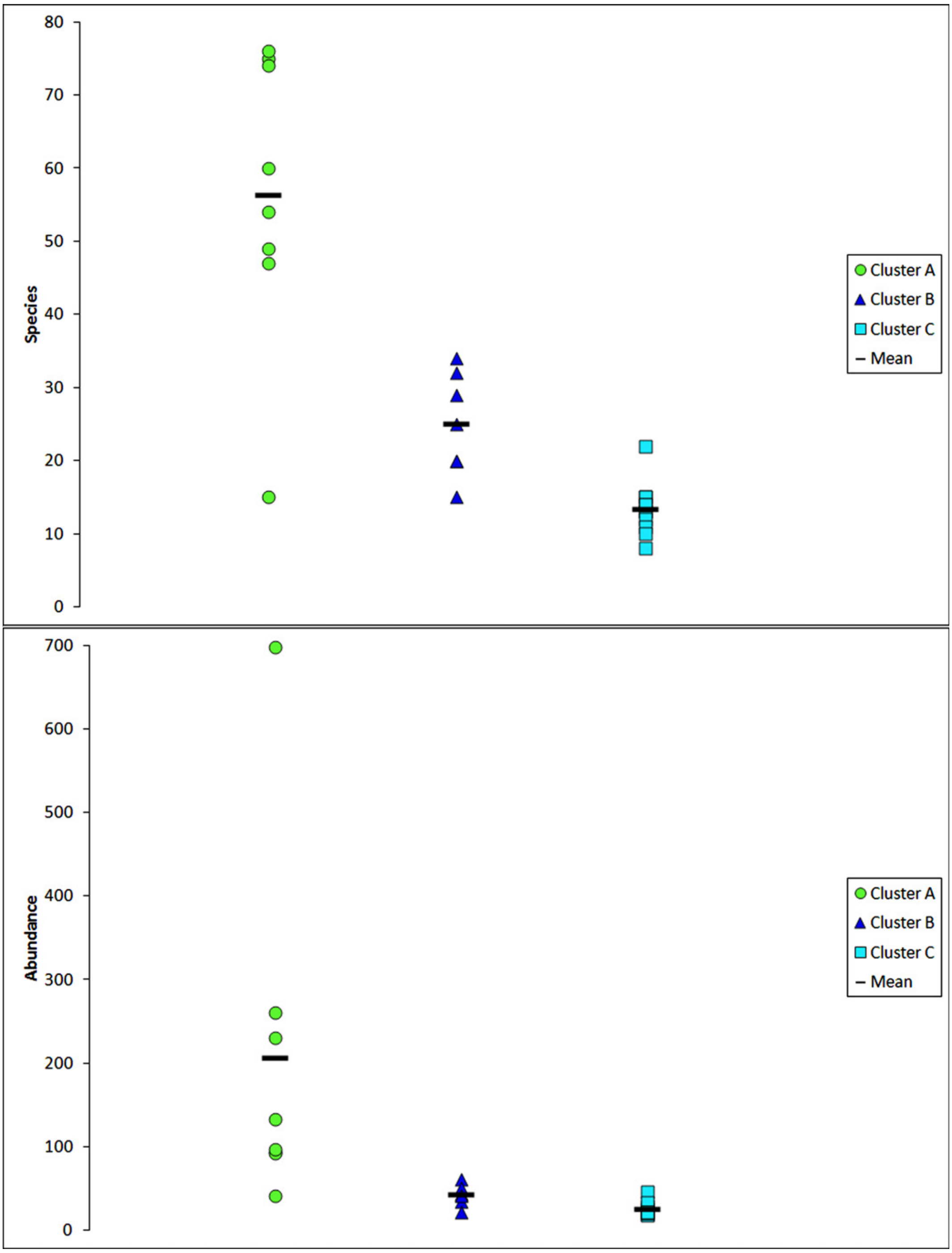

Fig. 3. Scatter graph showing the number of species (colonial taxa included) and individuals (colonial taxa excluded) present within samples from faunal cluster groups A, B and C.

The results of a series of 'Student's' two-sample *t*-tests to compare the mean proportion of % gravel, % sand and % silt/clay between the treatment groups (cluster group A, and dredged) showed all *p*-values were >0.05 (Table 4). The Null hypothesis was therefore accepted and it was concluded that there was no evidence of a statistically significant difference between treatment groups in terms of their percentages of gravel, sand and silt/clay. Whilst the results of Power analysis shown in Table 3 shows there was a greater than 0.8 probability of identifying a difference should one exist, it is important to reassess the results based on the pooled standard deviation (i.e. standard deviation associated with both treatment groups), particularly given the higher variability amongst dredged sediment samples seen in Fig. 5. Results of this analysis (Table 3, Power 2) show that the *t*-test comparisons have a statistical Power (*P*) of 0.72 (% gravel), 0.57 (% sand) 0.81 (% silt/clay) respectively. Overall, therefore, we can be reasonably confident that the lack of a difference identified between treatments is real, and that the composition of sediments within the dredged site at the time of survey were within the limits of normal variability associated with a comparable non-dredged area, thus establishing compliance with the hypothetical licence condition (stated above). However, by increasing the number of samples to 17, it would have been possible to achieve a statistical power of 0.9 or greater (Table 3, Power 3).

Table 1
Characterising species from each faunal cluster group (A–C) accounting for 50% of the within-cluster similarity as identified by a SIMPER analysis using square-root transformed macrofaunal abundance data (colonial taxa excluded).

Cluster A	Cluster B	Cluster C
Upogebia sp.	*Bathyporeia elegans*	*Notomastus sp.*
Lumbrineris gracilis	*Nephtys sp.*	*Nephtys sp.*
Poecilochaetus serpens	*Gastrosaccus spinifer*	*Echinocyamus pusillus*
Kurtiella bidentata		*Ophelia borealis*
Polycirrus sp.		*Aonides paucibranchiata*
Sabellaria spinulosa		*Polycirrus sp.*
Pomatoceros lamarcki		
Notomastus sp.		
Caulleriella alata		
Balanus crenatus		

This result was further strengthened by the findings of an ANOSIM test which showed that, based on the full multivariate particle size dataset, dredged samples were not statistically different from cluster group A (Table 2b). In addition, the test also showed that dredged sediments remained statistically significantly different from those belonging to cluster groups B and C.

4. Discussion

This study has shown that it is possible, using faunal assemblage data from the wider region, to identify limits for acceptable change in sediment particle size composition within an area likely to be affected by aggregate dredging. The approach utilises the fact that individual assemblages are naturally found in association with a range of different sediment particle size compositions. Therefore, as long as sediment composition within an impacted area remains within this defined range, it should be possible for the impacted community to recover after dredging has ceased, allowing for a period of natural recolonisation.

In this study, we identified three statistically distinct faunal groups in the region of the Hastings Shingle Bank. These included group A, a relatively species rich assemblage found on gravel dominated sediments, and groups B and C, which supported a relatively sparse and species poor faunal assemblage, typical of mobile sandy sediments. With reference to the distribution of faunal assemblages and the side scan sonar data, we made an assumption that prior to dredging the gravel dominated sediments within the active licence area were likely to have supported a similar assemblage to undisturbed gravel deposits adjacent to the licence area, namely group A.

A comparison of the sediment composition of samples taken from within the dredged area with those associated with faunal group A in the wider region showed that, in broad terms, sediment conditions within the extraction area remain within the limits of those found elsewhere for faunal group A. As such, were dredging to cease, it should be possible for faunal group A to return within the licence area. Evidence from Cooper et al. (2007b), who found faunal recovery within a relinquished part of the Hastings site after 7 years, supports this assertion.

Application of findings

With a proposed change in the way that certain licence conditions are set, it is important to consider the strengths and weaknesses of the proposed approach. In terms of strengths, it offers a number of advantages; namely:

1. It has a clear scientific rationale, by minimising the potential changes to the habitat (seabed sediments) it promotes the return of the original faunal assemblage to areas impacted by dredging. This approach will contribute to the long-term environmental sustainability of aggregate dredging, and help avoid potential cumulative effects.
2. Using the wider environment to set limits to change is an objective approach that takes account of local environmental conditions. This 'local issue' is important given the results in Cooper et al. (2011a), which showed that the sensitivity of faunal communities to a change in sediment particle size composition depends on the nature of the environment, a combination of sediment composition and the degree of natural physical disturbance. For this reason, limits to change are likely to be less stringent (less precise) in high energy sandy areas, and more stringent (more precise) in low energy gravelly areas.
3. Changes in the sediment composition, relative to a natural range identified during the EIA, are measurable and quantifiable, and will provide a real focus and renewed purpose to monitoring. The ability to measure change against some criterion will therefore facilitate the effective decision making on the part of the regulator when it comes to assessing the need for post-extraction environmental management, including an option for restoration (Cooper et al., 2011b, in press).
4. The proposed approach is pragmatic in the sense that it does not require the site to be returned to an exact representation of the pre-dredge state, an amount of change is expected and allowed for. It recognises that changes in sediment composition can occur, and that this is acceptable as long as changes are within the bounds of natural variability seen for the same faunal assemblage in the wider ecosystem.
5. The approach has the potential to reduce the cost of monitoring to the developer, by focusing efforts on assessing sediment composition. We believe that this could be achieved largely through remote acoustic means, reducing the need for costly benthic sampling surveys. Clearly there would still be a requirement for a pre-dredge benthic characterisation survey in order to identify whether there are important habitats and species within the area of potential effect, and to contribute to the setting of acceptable limits of change as described in the present study.

Whilst there are some obvious benefits to the suggested approach, some concerns remain. For example, the emphasis towards physical impact assessment at the expense of monitoring the benthos is likely to raise concern. Although significant effort goes into monitoring changes in the benthos during active dredging, it could be argued that these changes are to an extent predictable, particularly given recent advances in modelling, and the growing number of case studies highlighting the overriding importance of the physical environment in determining the structure and function of marine aggregate benthic communities (e.g. Shelton and Rolfe, 1972; Cressard, 1975; Millner et al., 1977; Bonvicini Pagliai et al., 1985; Lees et al., 1990; Kenny et al., 1998; Desprez, 2000; van Dalfsen et al., 2000; Boyd & Rees, 2003; Newell et al., 2004a,b; Sanchez-Moyano et al., 2004; Simonini et al., 2005; Cooper et al., 2007a,b; Moulaert and Kris Hostens, 2007; Cooper et al., 2008; Foden et al., 2009; Barrio Frojan et al., 2011; Last et al., 2011). It is also true that monitoring programmes are not always effective in detecting impacts, even when those impacts exist. This is due to the high number of samples required to detect change beyond the realms of natural variability, and the fact that the path of disturbance resulting from plume effects is not always known at the outset, despite best intentions. As we understand more about the impacts of aggregate dredging under a variety of different conditions, the need to monitor diminishes as impacts are increasingly predictable. That is not to say we have all the answers, merely that the remaining questions could be better addressed with a few intensive research sites, which can act as proxies for a large

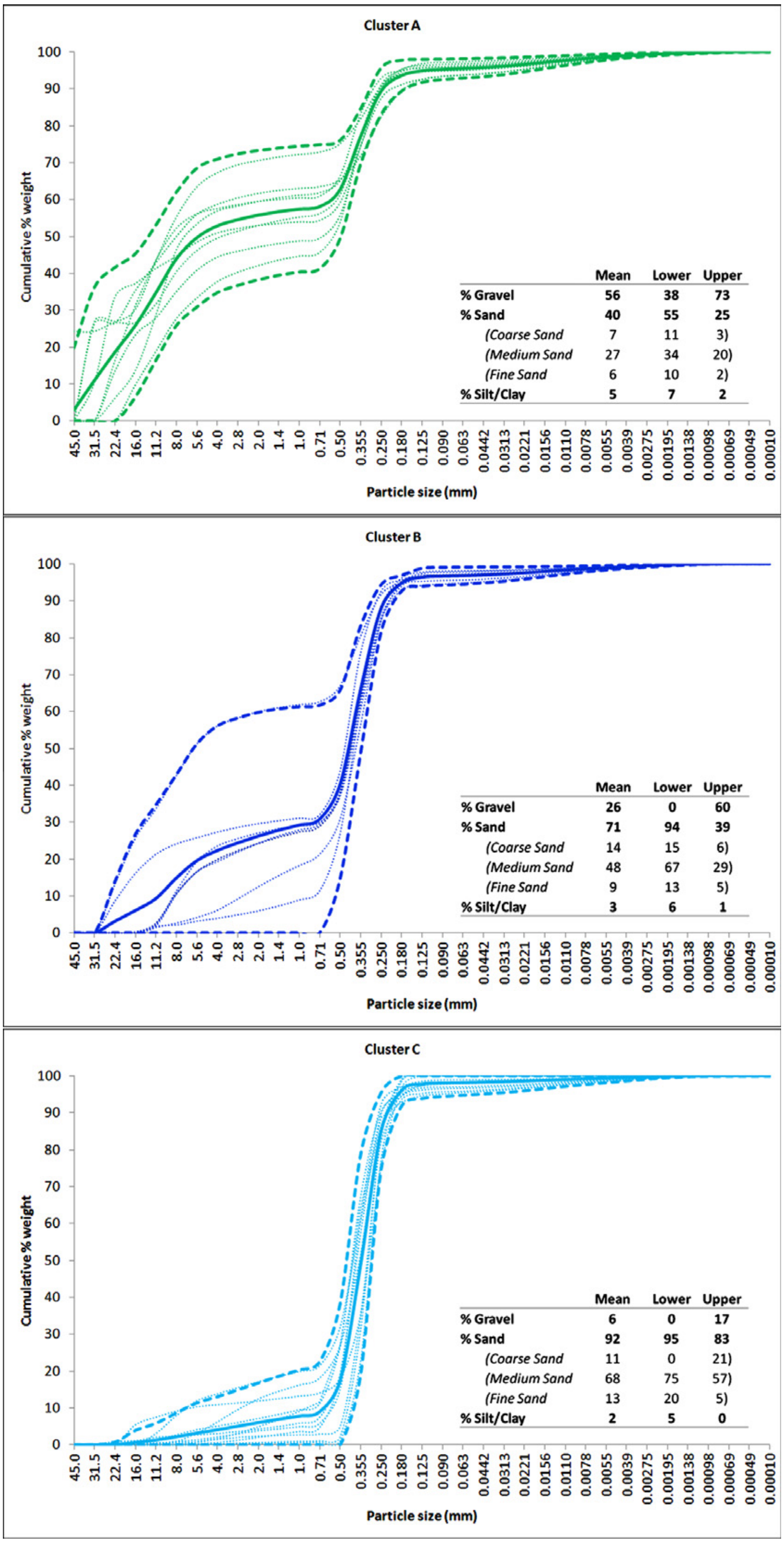

Fig. 4. Sediment particle size distribution plots for samples belonging to faunal cluster groups A–C. The solid and dashed lines show the mean and 95% confidence intervals (lower and upper) of the distribution. The inset table show the composition of sediments based on these three lines.

number of other extraction sites. The result of these and other existing research programmes will hopefully provide the necessary high quality evidence which can be incorporated into the EIA.

When a licence is granted, it is accepted that there will be changes to benthic fauna faunal assemblages within the licence boundary and possibly outside. The main concern of the regulator,

Table 2
Results of ANOSIM tests comparing (a) the sediment composition of samples assigned to cluster groups A–C, and (b) the sediment composition of dredged samples with the sediment composition of samples belonging to cluster groups A–C. Data used in this analysis were the non-transformed, full distribution sediment weight by 0.5 phi interval. A *p*-value of $\leqslant$0.05 shows a statistically significant difference between sample groups.

	Groups	*R* Statistic	*p*-value
(a)			
	A, B	0.354	0.005
	A, C	0.839	0.001
	B, C	0.203	0.05
(b)			
	Dredged, A	0.013	0.367
	Dredged, B	0.799	0.001
	Dredged, C	0.421	0.007

from a benthic ecology perspective, should be to ensure that impacts do not extend further than predicted, and that sites, after dredging, are left in a condition that does not hinder recovery to a pre-dredged condition. In terms of monitoring the effects of ongoing dredging, remote sensing methods may be particularly effective, allowing changes in sediment composition to be detected. Clearly, the extent of such surveys would need to go beyond the licence boundary in order to check for secondary effects. The acoustic surveys will also provide a vital check on results of particle size analysis. This is important as two sites can, in some circumstances, have a similar static particle size composition (as determined by *in situ* grab samples), yet the stratification of sediments on the bed can be very different due to bed-load sediment transport processes or the effects of sediment screening (Cooper et al., 2011b). The acoustic surveys could be supplemented with a number of regional monitoring stations to assess the condition of the biotopes in the wider region. In this way environmental managers can seek to regulate particular types of environmental pressures, possible resulting from different industry sectors, in order to maintain the ecosystem within a favourable state; this is surely the ultimate goal of 'marine spatial planning and the 'ecosystem approach' (CBD, 2000).

Table 4
Results of two-sample *t*-test comparing the mean proportions of major sediment fractions (% gravel, % sand, % silt/clay) between the dredged, and cluster group A treatment groups. Where *p*-value of the *t*-test is >0.05 this means there is insufficient evidence to reject the Null hypothesis (i.e. the means of the two groups are not statistically significantly different.

Sediment Class	$\bar{x}$ (Cluster A)		$\bar{x}$ (Dredged)		*p*-value
% Gravel	55.8	(±8.9)	52.3	(±16.3)	0.61
% Sand	39.7	(±9.4)	44.7	(±16.0)	0.46
% Silt/Clay	4.5	(±1.3)	2.9	(±1.7)	0.07

Table 3
Results of statistical Power analyses. Power 1 shows the power available to detect the required level of change $\bar{x}$ critical minus $\bar{x}$ for each sediment class based on the standard deviation (*s*) associated with sediments belonging to cluster group A. Power 2 shows the power available to detect the required level of change ($\bar{x}$ critical minus $\bar{x}$) based on the pooled standard deviation (*s*) associated with samples from both cluster group A and dredged treatments. Power 3 shows the number of samples necessary to detect the required level of change ($\bar{x}$ critical minus $\bar{x}$) with a minimum power of 0.9.

					Power 1		Power 2		Power 3		
Sediment class	$\bar{x}_{ClusterA}$	*n*	Change	α	*s* (Cluster A)	*P* (1 − β)	*s* (Pooled)	*P* (1 − β)	*s* (Pooled)	*P* (1 − β)	*n*
Gravel	55.8	8	±17.5	0.05	8.9	0.95	12.8	0.72	12.8	0.9	13
Sand	39.7	8	±14.8	0.05	9.4	0.84	12.9	0.57	12.9	0.9	17
Coarse	6.9	8	±4.0	0.05	2.6	0.86	–	–	–	–	–
Medium	26.6	8	±7.0	0.05	7.4	0.42	–	–	–	–	–
Fine	6.2	8	±4.0	0.05	2.5	0.79	–	–	–	–	–
Silt/Clay	4.5	8	±2.6	0.05	1.3	0.95	1.7	0.81	1.7	0.9	11

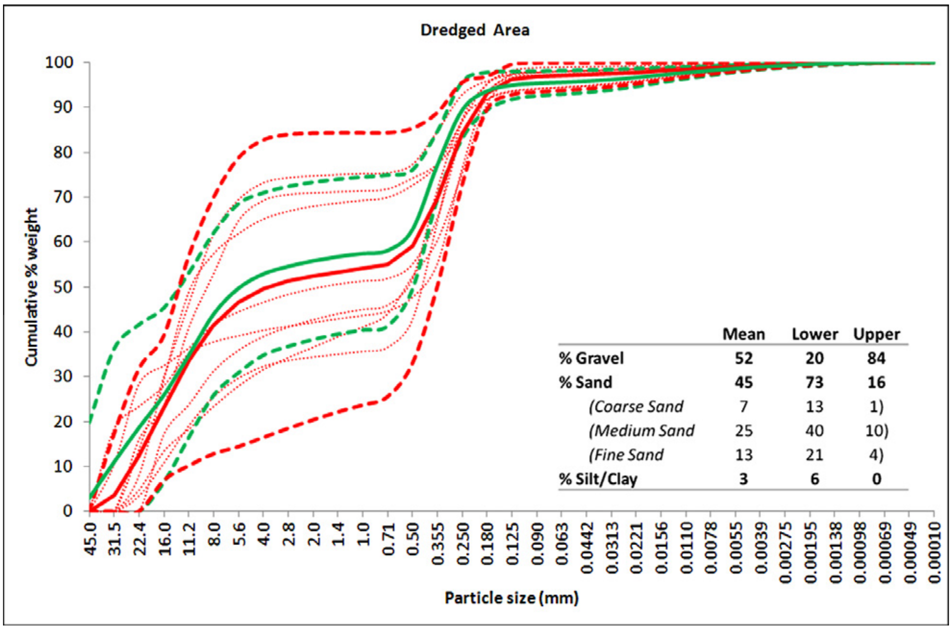

Fig. 5. Sediment particle size distribution for samples from the dredged area. The solid and dashed lines show the mean and 95% confidence intervals (lower and upper) limits of the distribution; the green lines are for the cluster group A in the wider region. The inset table show the composition of dredged sediments based on these three lines. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

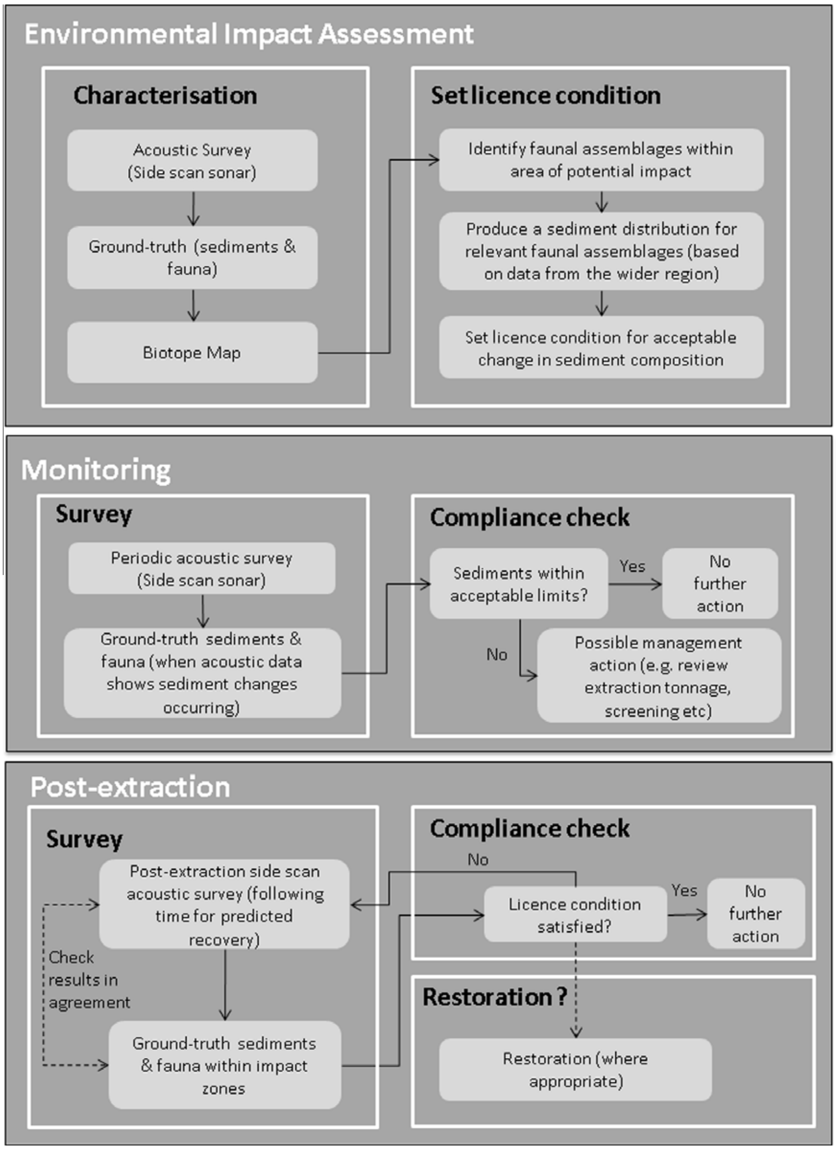

Fig. 6. Flow diagram outlining how the proposed new method for defining acceptable change in sediment composition might affect existing steps in the licensing and subsequent monitoring of the site.

A further concern relates to whether a change from one biotope to another is significant, particularly where one biotope is functionally equivalent to another (Cooper et al., 2008). In the case of this site, it is clear that a change from gravel to sand would result in a loss of species and individuals. Whether this is of detriment to the wider ecosystem is a difficult question to answer, although some progress, using modelling approaches, has been made recently (Kenny et al., 2010; Daskalov et al., 2011). In the absence of this understanding, then it is sensible to remain precautionary, as not to do so could lead to an incremental degradation of the environment. It is also sensible to consider the potential for perceived positive faunal changes associated with former extraction sites. For example, the development of *Sabellaria spinulosa* aggregations with formerly dredged areas of the present study site (Cooper et al., 2007b; Pearce et al., 2007). In such cases it may be appropriate to embrace the changes.

It is important to recognise that this study used data from an existing aggregate extraction site, and that assumptions were made concerning the likely pre-dredge status of the site. In my view this does not detract from the proposed method of assessment. However, it is clearly appropriate to undertake further testing of the approach, particularly in other regions where the sediment type and dynamics are known to be different, possibly in parallel with an existing monitoring programme. It is also important to recognise that the composition of resource deposits can vary with depth, alternating between sandier and more gravely sequences, possibly interspersed with lenses of finer sediment, including clays and silts (M. Russell, BMAPA, pers. comm., February 2012). It will be important to investigate whether the nature of such stratification might result in failure to meet the licence condition for acceptable change in sediment composition. If the proposed testing proves effective then the approach could be considered for use at other aggregate extraction sites.

Switching to the above method for determining the acceptability of changes in sediment composition would involve some changes in the current licensing and monitoring regime (see Ware and Kenny, 2011). A framework for the revised process is shown in Fig. 6.

Following the characterisation of the benthos, a licence condition for acceptable change in sediment particle size composition would be set. Monitoring, during the active phase of the licence would rely on acoustic data, the extent of which must include areas of secondary impact. Where obvious changes in the distribution of sediments are occurring, this would be followed up with a ground-truthing survey in order to collect samples of sediment for particle size analysis. It would be beneficial to report sediment data at a fine scale (% gravel, % coarse sand, % medium sand, % fine sand, % silt/clay), and to assess differences with the reference condition. In this way, even if the compliance with the licence condition is established at a coarse level, the data will provide useful information with which potential future problems can be identified (e.g. increases in medium sand associated with screening). This might allow for early management intervention, if necessary, to help ensure that the licence condition is met at the end of the licence term. Such action might include varying aggregate extraction rates, and limiting or allowing sediment screening. The frequency of monitoring could change according to emerging evidence from results, and future dredging plans.

At the end of the licence period a repeat of the characterisation survey would be undertaken, but this time covering only the area of dredging effect. Again, compliance with the condition for acceptable change in sediment composition would be established. If conditions were acceptable no further action would be required. If areas failed to meet the necessary pre-determined criteria, these would require additional monitoring, perhaps allowing additional time for natural recovery. Where compliance could not be achieved, despite allowing time for natural recovery, a decision regarding an option for restoration may be considered (Cooper et al., 2011b, in press; Collins and Mallinson, 2006). Where restoration is attempted, the aim would be to achieve compliance with the licence condition. Again, the licence condition would be extremely helpful in judging the success of restoration efforts.

In conclusion, the method presented here provides a scientifically justiable way of setting acceptance limits for change in sediment composition within areas impacted by marine aggregate dredging. In addition, as changes are measurable, this allows for objective decision making and transparency for all parties. The use of such limits could lead to lower costs to industry by switching the focus of monitoring onto the sediments, and will ensure of higher level of environmental protection than is offered at present.

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Impacts of physical disturbance on the recovery of a macrofaunal community: A comparative analysis using traditional and novel approaches

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ABSTRACT

The recovery of macrofaunal communities after marine aggregate dredging was assessed using both traditional indices (abundance, biomass and species diversity), and functional analysis techniques (Somatic Production, Taxonomic Distinctness, Infaunal Trophic Index, Biological Traits Analysis and Rao's Quadratic Entropy). A previously dredged area (Area 222), located off the southeast coast of England was selected for this investigation. Area 222 was split into sites that had been subjected to relatively high dredging intensity, relatively low dredging intensity, and undisturbed reference areas. Both traditional and functional analyses indicated that macrofauna at the low dredging intensity site had fully recovered at least 7 years after the dredging ceased. Recovery times at the high intensity site had a greater variability and most of the techniques recorded the recovery had yet to take place even 11 years after the dredging had ceased. Since Area 222 was dredged for a long period of time (approx. 25 years), it is suggested that a longer time series of study be carried out so that the definitive recovery period in this high intensity site can be determined. While a longer time series study is not always a realistic or cost effective, the present study could be useful to facilitate the selection of metrics to support in the assessment of macrofaunal recovery.

1. Introduction

Physical disturbance in the benthic environment can be from abiotic hydrodynamic processes that are responsible for sediment movement, bioturbation processes caused by the animals that live in or on the sediment, and disturbances caused by man's activities. Marine aggregate dredging (referred to as 'dredging' hereafter) is a common activity that is known to cause physical disturbance and produce detrimental impacts on the environment (Hall, 1994). The most common consequences from the repeated disruption of benthic environments are the disturbance of sediments, and degradation of associated fauna (Kaiser et al., 2006; Szymelfenig et al., 2006).

In England, a dredging 'permission' is granted by the Marine Management Organisation when the predicted impacts of proposed dredging are deemed acceptable. Production licences are subsequently issued by the Crown Estate, the owner of the UK seabed (MMO, 2010). Licence conditions are imposed in order to mitigate the effects of dredging locally and can include seasonal restrictions on dredging, limitations on dredging rate and restrictions on screening (the process of returning unwanted sediment fractions to the seabed) (DCLG, 2002).

Numerous studies have shown that on-going dredging is likely to reduce the number of species, abundance and biomass within dredged areas (e.g. Newell et al., 1998; Van Dalfsen et al., 2000). However, in comparison, relatively little is known about the longer-term consequences of dredging on the seabed (Boyd et al., 2003). In order to address this deficiency, the UK government has, over a number of years, funded research to address this issue (e.g. Boyd et al., 2004; Cooper et al., 2005, 2007; Gubbay, 2005). Many studies (e.g. Rees, 1987; Kenny, 1998) were carried out to assess the effect of the disturbance on dredged areas. However, most assessed effects were based on end points using biological and physical recovery data from other areas, and very few have produced definite recovery times (Cooper et al., 2007).

For many years, the effect of disturbance has been assessed using traditional metrics such as abundance, number of species and biomass (Bolam et al., 2006; Boyd et al., 2004; Cooper et al., 2005; Johnson and Frid, 1995; Newell et al., 1998). These assessments were carried out to determine the recovery of ecosystem, which is

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defined as having taken place when secondary succession returns the ecosystem to the pre-existing state (Borja et al., 2010). However, since screened dredging and the dynamic nature of the seabed alter the physical characteristics of the sediment, the original or pre-existing assemblage may no longer be able to be accommodated (Desprez, 2000; Matthews et al., 1996). Therefore, the use of traditional metrics should be treated with care when dealing with biological recovery as the assessment only incorporates the range and proportion of species present without taking into account the ecological and biological characteristics of the community. Mouillot et al. (2007) suggested that the extent to which species loss can alter basic ecosystem processes depends on the functional richness (i.e. the number of functional groups) in an ecosystem. In terms of dredging impact on functional diversity, communities of organisms inhabiting an area of dredged seabed possibly differ in composition or diversity from the pre-dredged state, but may develop similar functional capacity through the recovery process. Therefore, system recovery may not require similar biomass, biodiversity or community composition. This is due to possible functional redundancy, whereby the loss of a particular species does not affect ecosystem function since the function performed by that species is taken up by another species from the same functional group (Lévêque and Mounolou, 2003; Walker, 1992). To address this issue, many studies have recently focussed on functional diversity to assess faunal recovery following anthropogenic perturbations by incorporating biological differences among species (e.g. Borja et al., 2000; Botta-Dukat, 2005; Bremner et al., 2006a,b; Maurer et al., 1999; Josefson et al., 2009). Biological difference, which can be drawn from functional traits, is proved to respond significantly to human disturbance. Function- or trait-based diversity metrics may thus represent appropriate additional methods for assessing changes in ecosystem function (Péru and Dolédec, 2010).

The present study aims to investigate the recovery of the macrofaunal community using two different approaches: (1) traditional method; where the recovery was assessed by simply looking at the macrofaunal assemblage composition, and (2) functional method; where recovery is assessed in relation to the functional capacity (or health) of the ecosystem. By understanding how different metrics work, this study can be used as an initial baseline in selecting suitable metrics to be applied for specific study on ecosystem recovery. Since there is a strong link between ecosystem processes and functional traits, the use of functional metrics in addition to the traditional ones may facilitate the assessment of ecosystem functioning (e.g. Charvet et al., 2000; Dolédec and Statzner, 2008; Gayraud et al., 2003).

2. Materials and methods

2.1. Study site

Area 222 (Fig. 1) is located approximately 20 miles to the east of Felixstowe, off the southeast coast of England. Water depths range between 27 m and 35 m Lowest Astronomical Tide (LAT). The site was first licensed for dredging in 1971 with approximately 0.3 km^2 of dredging area. The dredging work reached its peak in 1974 with 872,000 t of aggregate removed per annum. From 1975 to 1995, dredging continued at lower levels but still exceeded 100,000 t per annum. The last dredging activity took place in 1996 when approximately 12,000 t of aggregate was removed (Boyd et al., 2004; Cooper et al., 2005). According to Boyd et al. (2004), this area was subjected to screening processes which altered the sand:gravel ratios. Dredging was carried out in this area and its vicinity using trailer and static suction hopper, which creates a track of approximately 2.5 m wide from its movement over the seabed.

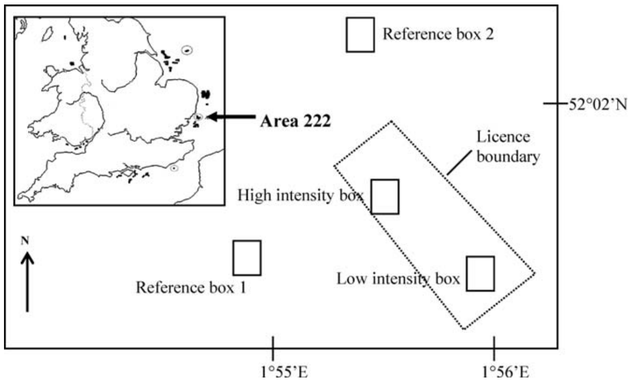

Fig. 1. The location of Area 222 dredging licenses and sample locations.

2.2. Field sampling and measurements

Since 1993, an Electronic Monitoring System (EMS) has been fitted to all dredging vessels on a Crown Estate licence (Boyd et al., 2004; Cooper et al., 2005). Information obtained from EMS was used in order to determine the dredging intensity of different areas. The selected sampling areas were separated into high dredging intensity (referred to as High hereafter: >10 h of dredging within a 100 m by 100 m block during 1995), low intensity (referred to as Low hereafter: <1 h of dredging within a 100 m by 100 m block during 1995), and two undisturbed (referred to as Reference/Ref hereafter) sites (Boyd et al., 2004; Cooper et al., 2005). All sampling works were carried out from 2001 to 2002 using RV Cirolana and from 2003 to 2004 and 2007 using RV Cefas Endeavour. In order to avoid any significant seasonal effects, all fieldwork was undertaken at the same time of year, between May and July.

2.3. Sample collection and processing

Samples were collected using a 0.1 m^2 Hamon Grab. Ten replicate samples were randomly taken within High and Low sites, and 5 replicate samples were collected at both Reference sites, totalling 30 samples per year. A 500 ml sub-sample was removed from the grab for sediment particle size analysis. The remaining sample was then washed over 5 mm and 1 mm square mesh sieves to remove the fine sediment. The sediment remaining on both screens was then back-washed into a sealable container and fixed in 4–6% buffered Formaldehyde solution (diluted in seawater). Samples were taken to the laboratory and were processed according to standard methodology (see DTLR, 2002 for details of sample processing). Specimens were identified to the lowest possible taxonomic level. For biomass measurements, each taxon in every sample was blotted on absorbent paper before being weighed (wet weight) to the nearest 0.0001 g. The measured wet weight was converted to ash free dry weights (AFDW) using standard conversion factors (Ricciardi and Bourget, 1998).

2.4. Traditional statistical analyses

Boyd et al. (2004) and Cooper et al. (2005) analysed dredge data from 2001 to 2004 using univariate measures such as total abundance, species richness and biomass. In this present study, the same analyses were applied but using an additional set of data from 2007. Further diversity analyses were also carried out: Species richness (S) and Margalef index (Dm) calculated the richness while both Simpson (complement) index (Ds) and Pielou's index (J') are based on the evenness of the community (Magurran, 2004).

2.5. Functional statistical analyses

Cooper et al. (2008) reviewed 12 functional analyses to quantify functional diversity and recommended 5 techniques as being suitable for use with benthic macrofaunal data collected from the Hastings Shingle Bank. Considering the similarity of the nature of study and the study area, the same techniques were also applied in the present study. The techniques were Somatic Production, Taxonomic Distinctness, Infaunal Trophic Index, Biological Traits Analysis and Rao's Quadratic Entropy. A more detailed calculation of these metrics is available in Cooper et al. (2008).

2.5.1. Somatic Production

In terms of energy flow, Brey (2001) defined Somatic Production (P_s) as the part of the food consumption processes that change the biomass (B) of an organism with time, and subsequently would potentially be available as food for other organisms in the next trophic level. Production to biomass ratio (P_s/B) is generally used to describe the turnover of a population. This ratio is mainly affected by life history characteristics such as density, recruitment, age, life span (Cusson, 2005), and abiotic factors such temperature and depth (Brey and Clarke, 1993).

2.5.2. Taxonomic Distinctness

Although it does not directly signify functional diversity, Taxonomic Distinctness (TD) has been used as a proxy in this sense and can be calculated using the PRIMER 6 package (Clarke and Warwick, 2001b). This is based on the understanding that a community is more diverse if it comprises species that arise from different genera or higher taxonomic levels as compared to a community with closely related species (Clarke and Warwick, 1998, 2001a; Magurran, 2004; Somerfield et al., 2008), and a more diverse community is likely to be capable of performing more functions (Cooper et al., 2008). TD measures the average taxonomic distance, which is the path length between two randomly chosen species, traced through the taxonomic classification in an assemblage (Clarke and Warwick, 1998).

2.5.3. Infaunal Trophic Index

Infaunal Trophic Index (ITI) is a more targeted approach to monitor the response of a marine environment to organic enrichment or flux. This technique measures the community response to available organic material based on the dominant feeding mechanisms (Maurer et al., 1999). An organisms' feeding type is considered to be one of the central processes where ecosystem function is expressed (Pearson and Rosenberg, 1987). ITI values ranging from 80 to 100 indicate a reference condition. Values from 60 to 80 indicate normal/unaffected condition, values from 30 to 60 represent a modified condition and values from 0 to 30 indicate a degraded or polluted condition (Maurer et al., 1999; Word, 1979).

2.5.4. Biological Traits Analysis

Biological Traits Analysis (BTA) uses specific species traits and variation in the pattern of traits of biological components to assess the functioning of ecosystems (Bremner et al., 2006b; Bremner, 2008; Marchini et al., 2008). As habitat variability controls the community structure, knowing the community's biological characteristics provides information on how the organisms respond to stress, and therefore the functional diversity status can be identified. Biological traits, which are directly related to ecosystem structuring mechanisms, are able to directly illustrate the factors that drive the change in communities (Bremner et al., 2003). For the present investigation, a species-by-traits matrix was produced based on eight traits: Size, Larval type, Relative adult mobility, Body form, Degree of attachment, Adult life habit, Feeding habit and Habitat. Each of these traits contained several different categories; and species were assigned to these categories using fuzzy coding. For example, the trait Larval type was divided into 3 categories namely Planktotroph, Lecitotroph and Direct development. Biological Traits Analysis (BTA) does not produce a value which can be analysed with the univariate analyses chosen for this study; hence the trait-by-sample values of this technique were analysed using multivariate analyses to compare the dispersion of assemblages between sites.

2.5.5. Rao's Quadratic Entropy

Botta-Dukat (2005) proposed a functional diversity index based on quadratic entropy (Rao, 1982) which measures the pair-wise distance of functional differences between species. This index, FD_Q also incorporates the number of individuals present in the community. FD_Q is a generalised form of the Simpson diversity index and in its calculation, both diversity and dissimilarity elements are addressed (Petchey and Gaston, 2002; Mason et al., 2003). Rao's Quadratic Entropy satisfies *a priori* criteria to be a suitable functional diversity index as it (1) utilises more than one trait, and (2) incorporates species abundance; making it able to treat functional types differently based on the abundance value (Botta-Dukat, 2005).

2.6. Multivariate analyses

The PRIMER 6 package was used for all multivariate analyses. Non-parametric multi-dimensional scaling (MDS) ordination using the Bray-Curtis similarity measure was used to show the dispersion of samples between sites. Analysis of Similarities (ANOSIM) was performed to generate a test statistic R that provides values between 0 (when no differences between assemblages are recorded) and 1 (where the assemblages are completely different). The nature of the community groupings identified in the MDS ordinations was explored further by applying the similarity percentages program (SIMPER) to determine the taxa or functional traits which contributed most substantially between different sites.

3. Results

3.1. Sediments

Both Low and Reference sites recorded fairly similar sediment characteristics with the highest proportion made up of gravel, as well as medium sand at the Low sites and silt/clay at the Reference sites (Fig. 2). At the High site, coarse sand dominated in 2001 and 2002, but this decreased over time to become comparable to Low and Reference sites, while the proportion of gravel was less in 2001 and 2002, but increasing in proportion to become comparable to Low and Reference sites by 2004 and 2007.

3.2. Macrofauna – traditional analysis

3.2.1. Abundance and biomass

Throughout the study period, the macrofaunal assemblage found within the Low site recorded some degree of similarity with the Reference site in both the number of individuals (N) and the biomass (AFDW) (Fig. 3). The abundance was significantly lower ($p < 0.05$) at the High site than the Reference site from 2001 to 2004, but by 2007 the difference was not significant ($p > 0.05$). Macrofaunal assemblage also has a strong correlation with the dominant grain sizes. The assemblage recorded a positive correlation (Spearman's rank correlation: $r_s = 0.593$) with gravel, while negatively correlated with sand ($r_s = -0.632$). The biomass at the High site was significantly lower ($p < 0.05$) than the Reference site in 2002 and 2007. Multidimensional scaling (MDS) shows samples from the High site are relatively widely dispersed in comparison with

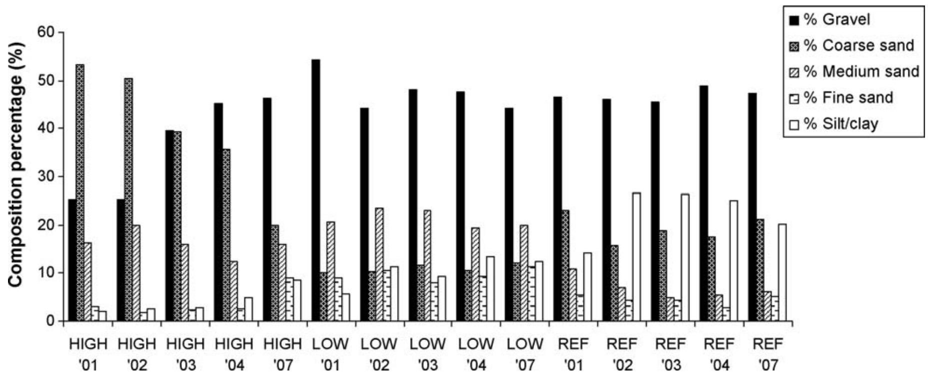

Fig. 2. Percentage of sediment particles composition at High and Low dredging intensity sites and at Reference site from 2001 to 2007.

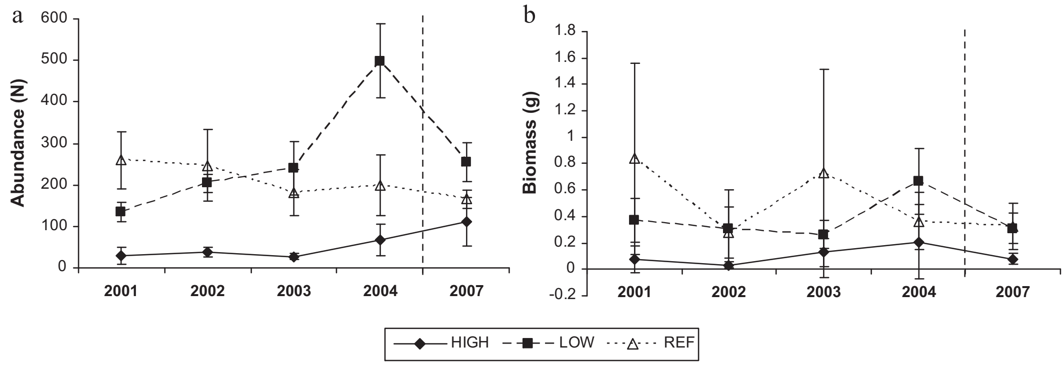

Fig. 3. Summary of means and 95% confidence intervals of (a) number of individuals and (b) biomass (AFDW) at sites of High and Low dredging intensity and Reference site. The dotted vertical line indicates the discrepancy in time intervals.

the tighter clustering of samples from the Low and Reference sites (Fig. 4).

3.2.2. Species diversity

The High site had significantly lower ($p < 0.05$) Species richness (S) and Margalef (Dm) compared to the Reference site from 2001 to 2007, whereas the Low site recorded no difference (Fig. 5a and b). There was no significant difference ($p > 0.05$) in Simpson's (Ds) and Pielou's (J') between High and Reference sites by 2001 and 2004 respectively. The Low site in 2004 however recorded significantly lower ($p < 0.05$) values than the Ref site for both indices (Fig. 5c and d). This might be due to a high abundance of *Pomatoceros lamarcki* that reduced the values of these indices which are based on dominance.

3.3. Macrofauna – functional analysis

3.3.1. Somatic Production

Production values (P_s) of the High site are more spatially scattered than those of the Low and Reference sites. A tighter cluster of samples for Low and Reference sites indicates the similarity between these sites (Fig. 6), and this is confirmed by the ANOSIM values (Table 1). There was no significant difference ($p > 0.05$) in the level of production at the Low site compared to the Reference sites in 2002.

3.3.2. Taxonomic Distinctness (TD)

TD values in dredging sites were consistently lower than the Reference site throughout the study period, but were only signifi-

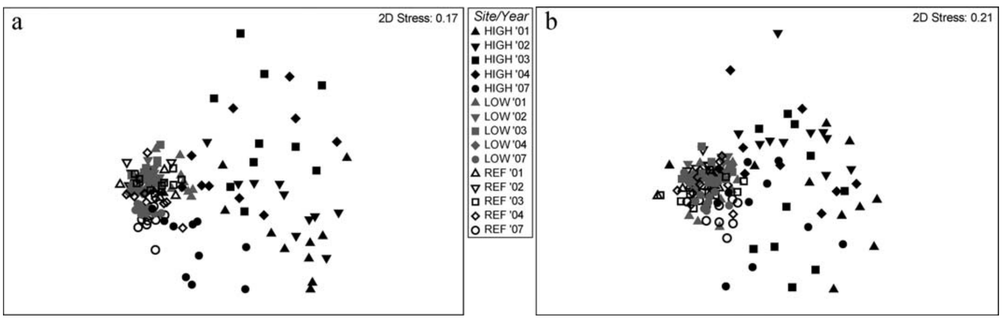

Fig. 4. An MDS plot of Bray-Curtis similarity based on square root transformed data of (a) Number of individuals and (b) Biomass.

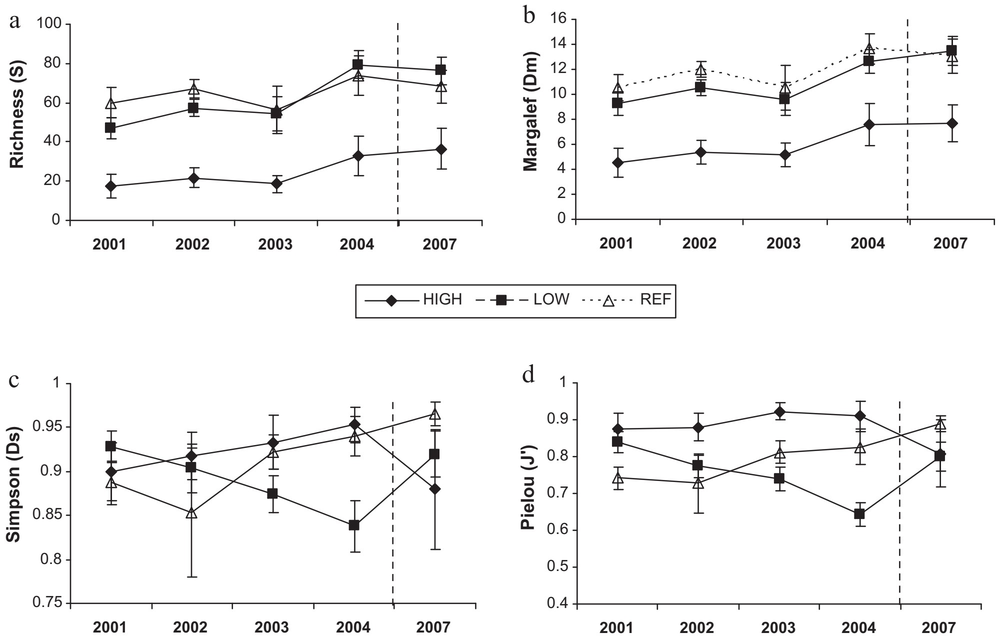

Fig. 5. Mean values (±95% confidence intervals) of (a) Species richness, (b) Margalef index, (c) Simpson (complement) index and (d) Pielou index at High and Low intensity dredging sites and Reference site from 2001 to 2007. The dotted vertical line indicates the discrepancy in time intervals.

cantly lower ($p<0.05$) in the 2001 (High site) and 2007 (Low and High sites) (Fig. 7a). Samples from the High site are skewed toward the left hand side of the funnel plot indicating the lower number of species in this site compared to both Low and Reference sites (Fig. 8). Although there are several samples grouped outside the funnel lines, the grouping of most of the samples (including the High site) inside the lines suggests that there are no significant change of the taxonomic distinctness.

3.3.3. *Infaunal Trophic Index (ITI)*

ITI values from 2001 to 2007 ranged from 62 to 84 (Fig. 7b). While mean values from any of the sites rarely exceeded 80 (a reference condition), those of both the High and the Low sites were more consistently higher than 60, indicating that they were at least in a normal (or unaffected) condition throughout the study period. High site samples on the MDS plot are widely dispersed from the samples of the Low and Reference sites (Fig. 9a), and this observation is confirmed by ANOSIM values where higher *R* values were recorded in the High site (Table 1).

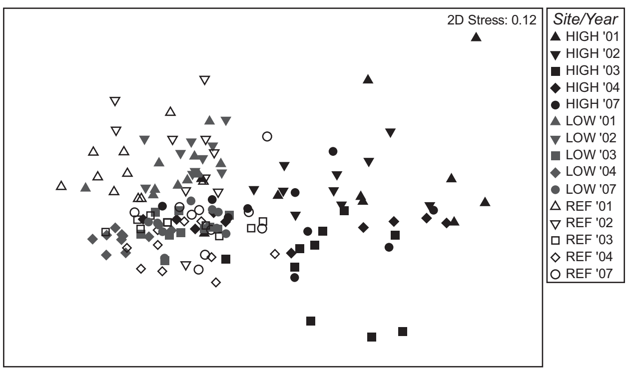

Fig. 6. An MDS plot of Bray-Curtis similarity of Somatic Production at High and Low intensity dredging sites and Reference site from 2001 to 2007.

3.3.4. *Biological Traits Analysis*

Samples from the Low site were closely grouped with the samples from the Reference site (Fig. 9b). The biological traits of the community at the High site were significantly different compared to the Reference site (Table 1). No difference was recorded at the Low site by 2003, although one year after that (2004) and in 2007 the biological functions of the assemblages differ from the Reference site. The 2003, 2004 and 2007 Low intensity dredging site results were analysed using SIMPER analysis. In those years, the top two trait categories that contributed the most to the dissimilarity between the two sites were the 'Planktotroph' and '1–3 cm', respectively from traits Larval type and Size. Further investigation through SIMPER analysis of abundance data revealed that the considerably high abundance of *Pomatoceros lamarcki* in 2004 and 2007 contributed to the functional difference in these sites compared to

Table 1
Summary of *R*-values derived from ANOSIM test based on functional techniques.

Technique	High/Ref	Low/Ref
P_s		
2001	0.685**	0.164*
2002	0.742**	−0.005
2003	0.555**	0.025
2004	0.229**	0.282**
2007	0.145*	−0.030
ITI		
2001	0.843**	0.540**
2002	0.943**	0.284**
2003	0.882**	0.036
2004	0.410**	0.479**
2007	0.369**	0.282**
BTA		
2001	0.880**	0.533**
2002	0.953**	0.122*
2003	0.908**	0.002
2004	0.353**	0.581**
2007	0.385**	0.400**

* Significant difference at $p<0.05$.
** Significant different at $p<0.01$.

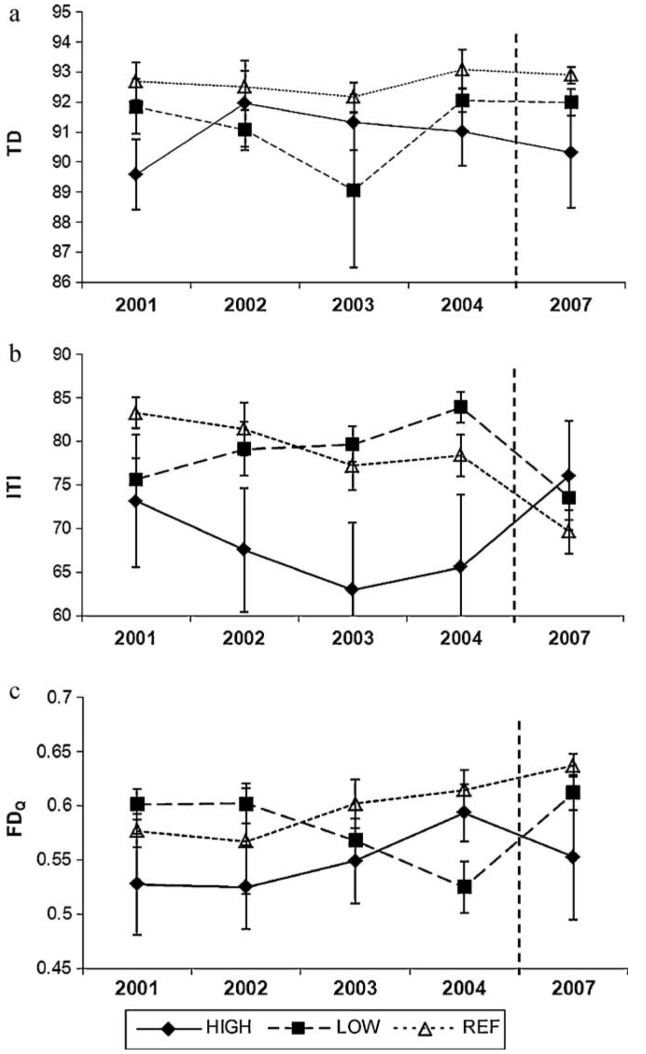

Fig. 7. Annual mean values (±95% confidence intervals) of (a) Taxonomic Distinctness, (b) Infaunal Trophic Index and (c) Rao's Quadratic Entropy for High and Low intensity dredging sites and Reference site. The dotted vertical line indicates the discrepancy in time intervals.

the Reference site. This polychaete worm, which has 100 per cent affinity to the two categories, was the highest contributor to the dissimilarity between these sites in 2004 and 2007, but not in 2003 where the highest contributor was *Anthura gracilis*.

3.3.5. *Rao's Quadratic Entropy*

The average value of Rao's Quadratic Entropy at the High site was generally lower than that recorded at Low and Reference sites (Fig. 7c). The difference however was not significant ($p > 0.05$) throughout the study period. The only significant difference ($p < 0.05$) was recorded at the Low site in 2004 and the High site in 2007. This might be due to the strange property of this index where its value may decrease as a consequence of an increasing number of species (Botta-Dukat, 2005). The reason is that FD_Q is influenced by both species-abundance based diversity and by trait differences among species. As in the case of the Low site in 2004 and at the High site in 2007, high number of individuals had increased species-abundance based diversity, but at the same time it may decrease the average dissimilarity among species as the addition of new individuals did not proportionally add new traits into the system. The decrease of average dissimilarity means these sites have a high average similarity, and hence a smaller functional diversity (Botta-Dukat, 2005).

4. Discussion

Biological impacts of dredging on macrofaunal communities are usually associated with a reduction in the number of taxa, abundance and biomass (Newell et al., 1998) as well as changing the population and community composition (Sánchez-Moyano et al., 2004). There are several factors which affect this process including the community type of disturbed and unaffected areas (Van Dalfsen et al., 2000), dredging intensity and penetration into substratum (Kaiser and Spencer, 1996), life cycles and feeding strategies (Lopez-Jamar et al., 1986) and the settlement of larvae and immigration of mobile species (Hall, 1994).

Macrofaunal communities in this study recorded a faster recovery (or at least progress toward recovery) at the Low site than at the High site. In the case of the High site, this might be due to a greater shift of sediment composition as a result of greater dredging intensity; hence it needed a longer period to return to its original state. This site had a finer sediment particle size (i.e. lower percentage of gravel and higher percentage of coarse sand) compared to Low and Reference sites during the early period of investigation, but the

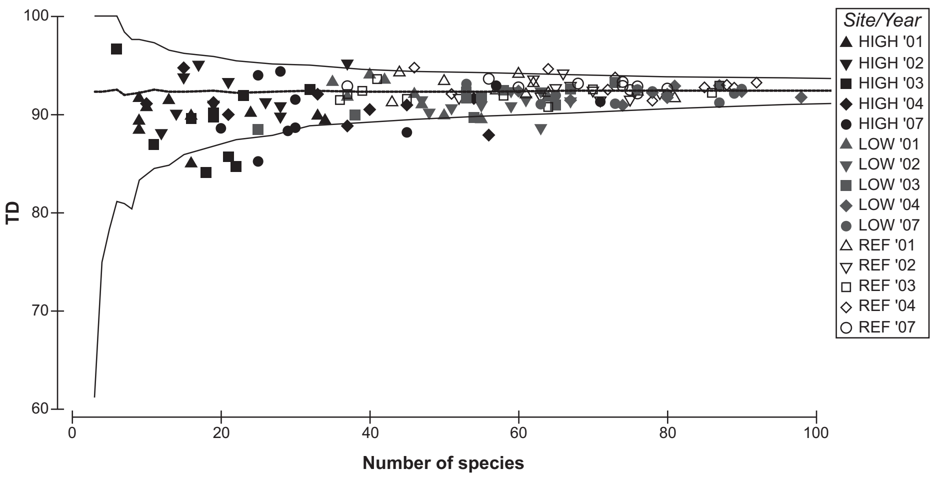

Fig. 8. Funnel plot for Taxonomic Distinctness (TD) of High and Low intensity sites and Reference site. The funnel plot graphs the distribution of assemblages based on the number of species in every station. The mean TD of the whole assemblage is represented by the dotted horizontal line, while the funnel lines indicate the 95% confidence limits.

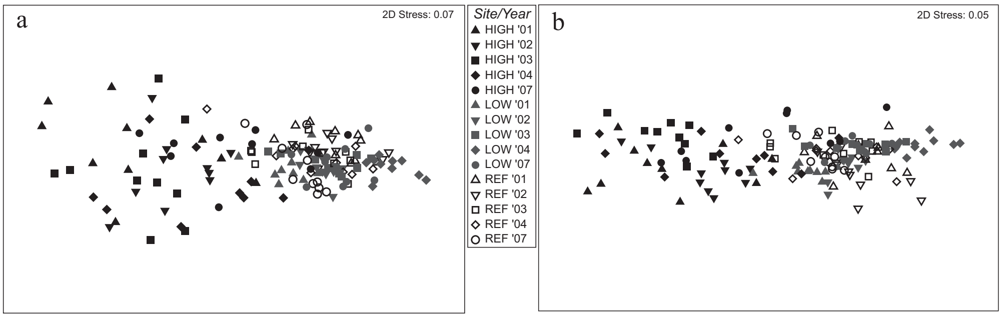

Fig. 9. An MDS plot of Bray-Curtis similarity of (a) Infaunal Trophic Index and (b) Biological Traits Analysis at High and Low intensity dredging sites and Reference site from 2001 to 2007.

sediment proportions became fairly similar to both Low and Reference sites from 2004 to 2007 (Fig. 2). At the Low site, the percentage of gravel (which had a positive correlation with the assemblage) had returned to the condition as in the Reference site before this study commenced, and this early physical recovery has promoted an early biological recovery. Conversely, the High site recorded a low percentage of gravel, but higher percentage of coarse sand as compared to the Reference site. As this proportion is now becoming similar (2004 and 2007), it might be suggested that physical recovery was almost complete in 2007, so that biologically recovery can now take place over the next few years.

The period of time needed for system recovery varies depending on the method of assessment used. The recovery at the High site recorded varied timescales using both traditional and functional methods, with differences of more than 5 years (Table 2). Recovery times were more consistent at the Low site, with only one to two years difference between all the analyses. Traditional analysis recorded a more consistent result with all except *N* recording the recovery at the Low site as early as 2001, or five years after the dredging terminated. Functional analyses recorded rather different recovery times, with ITI and BTA, which are both based on functional traits recording a similar time. Despite the consistency showed by traditional analyses, it might be unwise to suggest this technique is more suitable than functional techniques. This consistency might be due to the traditional analyses (*S*, *Dm*, *Ds*, *J′*) using a similar mathematical basis (i.e. number of species). Therefore it is unsurprising if these analyses produced a similar recovery period as opposed to functional analyses, which use a different basis, and hence produced different recovery times.

The period of time (between <5 to >11 years) needed for recovery in the present study differs from other studies involving physical disturbance (e.g. Borja et al., 2009; Powilleit et al., 2006; Simonini et al., 2007; Wilber et al., 2007), which in general took only 2–4 years. However, these studies were conducted in areas with a one-off disturbance with no 'legacy' stressor (Borja et al., 2010). In contrast, Area 222 in the present study was continuously dredged for 25 years; hence the physical change of the seabed was expected to be seen. For example, there is evidence of dredge depressions resulting from static suction hopper dredger (Cooper et al., 2005). This type of impact is likely to prolong the recovery of the macrofaunal communities. Elliott et al. (2007) pointed out that the trajectory of degradation may be different from the trajectory of recovery. Therefore in the case of the present study, the time taken for it to recover could be longer from the period of the disturbance. Foden et al. (2009) compared recovery rates of dredging areas around UK and they found out that a faster recovery occurred in areas with strong tidal stress and highly mobile sands. Physical features in Area 222, which characterised by weak-moderate tidal stress and medium/coarse sand are believed to extend the recovery in present study.

Table 2
Recovery times at the High and Low dredging intensity sites based on the different analyses tested.

Index	Year of recovery (number of year after dredging)	
	Low intensity site	High intensity site
Abundance (*N*)	2002 (6)	2007 (11)
Biomass-AFDW (*B*)	≤2001 (≤5)	>2007 (>11)
Richness (*S*)	≤2001 (≤5)	>2007 (>11)
Margalef (*Dm*)	≤2001 (≤5)	>2007 (>11)
Simpson (*Ds*)	≤2001 (≤5)	≤2001 (≤5)
Pielou (*J′*)	2002 (6)	2004 (8)
Somatic Production (P_s)	2002 (6)	>2007 (>11)
Taxonomic Distinctness (TD)	≤2001 (≤5)	2002 (6)
Infaunal Trophic Index (ITI)	2003 (7)	>2007 (>11)
Biological Traits Analysis (BTA)	2003 (7)	>2007 (>11)
Rao's Quadratic Entropy (FD_Q)	≤2001 (≤5)	≤2001 (≤5)

4.1. Index comparison

While the recovery times suggested by different analyses were varied, some degree of overlapping of samples in MDS ordinations suggests that the faunal assemblage had nearly recovered. Most analyses except P_s, ITI, BTA, *J′* and *N* all suggested that recovery at the Low site had already taken place prior to the start of this study. The fastest recovery at the High site is indicated by *Ds* and FD_Q 5 years after dredging ceased. The similarity between these techniques might be due to the fact that FD_Q is a generalised form of *Ds* (Lepš et al., 2006).

In comparison of the functional metrics, P_s is highly relevant to understanding ecosystem function (Pombo et al., 2007). However, given that conversion factors for production are currently only available at the family level there is a potential for inaccurate results. For example, individual species within a family may in fact exhibit different rates of production. For this reason we must be cautious in assuming that there are indeed no differences in production between the sites in the present study.

In contrast to production, TD might give a more detailed insight to assessing functional diversity as its measurement is based on the species level data. Also, it is perceived to be robust and is not affected by sampling effect since no abundance data are incorporated. Nevertheless, the measurement only considers the species on an equal basis while ignoring the effects of different abundances.

Feeding behaviour, which is the central process in an ecosystem, is used as the basis for the ITI. Intrinsically, feeding behaviour can be used to measure the community response to organic materials. The change in dominance of organisms from suspension feeder to deposit feeder may indicate an increase in the amount of sediment particulate organic matter (Maurer et al., 1999). The drawback of this index is due to its focus only on feeding behaviour. As a result, the ITI may thus be less sensitive in detecting disturbance. For instance this study shows all the sites are in 'normal' condition despite aspects of community structure (abundance, biomass and richness) all showing the High site to be still impacted.

BTA has an advantage for determining ecosystem function because this index uses multiple functional traits. As the traits reflect the function of every species in a community, the BTA is useful to show the link between organisms and their environment (Bremner et al., 2006b). However, there are limitations of this metric that need to be taken into account. Given there is a wide range of traits available, selection of a few most important and meaningful traits can be problematic. Likewise, dismissing certain traits could give false interpretation on describing the relationship within assemblages especially if the traits have the biggest influence on the whole community (Bremner et al., 2006b).

The same considerations of the (dis)advantages of multiple traits should also be given if FD_Q were to be selected. However FD_Q seems to be more detail as it measures the pair-wise dissimilarity between every species in a sample, compared to BTA that simply sums up the traits values of all species in the sample. As mentioned above, great care should be given to the results of this technique given its unexpected property where values may decrease if species richness increases. However, this downside might be mitigated when comparing this index at a large spatial scale given the fact that speciose communities normally correspond to high abundance and high diversity taxa (Péru and Dolédec, 2010). Therefore the unexpected property could be avoided as the increase of abundance is proportional to the increase of species richness.

For managerial purposes, it is impractical to use all the indices to determine the recovery of an area after disturbance. As every index uses different calculations and gives different interpretations, the selection of indices or techniques should be made to reflect the purpose of the study. For example, the desired recovery could be on the basis of community structure or a fully functioning ecosystem. An investigation about the impact of disturbance on the variety of species might suitably be based on species diversity indices (i.e. *S*, *Dm*, *Ds*, J') and TD. In this case however, TD gives a better interpretation of diversity as it takes into account the higher taxonomic level (e.g. family and class). The use of P_s and *B* in a study is useful to assess the potential availability of food resources (e.g. for the fishing industry), as these indices are based on the body size of the macrofauna. ITI, which focuses exclusively on feeding guilds, could be used to determine the change of organic enrichment in a system. If the study of habitat restoration is of interest, analysis using BTA and FD_Q might be more suitable.

5. Conclusion

In assessing the recovery of functional diversity, it is vital to use rigorous statistical techniques so that the dissemination of the output to the government, scientific community, industrial stakeholders and general public can be adequately achieved. This study was conducted using several techniques with different characteristics, which in return produced different results. It is therefore unwise to make a judgement of which technique is better than the others because they have different suitability to different purposes of study. Although functional analysis merits more attention and consideration, it should be used in conjunction with the traditional metrics. The variability of the outcome from both types of technique also suggests that further investigation should be conducted for a longer time series, perhaps to compensate for the long period of dredging in this area (1971–1996). This is especially true of the High site where most of the indices indicate that recovery is still on-going 11 years after removing the stressor. In addition, a longer time series study will minimise the possibility of detecting changes that are simply associated with natural variability (Hewitt et al., 2001). Nevertheless, given that the extension of the time-series at Area 222 may not be possible due to financial constraints, this study will be useful in facilitating the selection of metrics to support in the assessment of macrofaunal recovery following disturbance.

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Implications of dredging induced changes in sediment particle size composition for the structure and function of marine benthic macrofaunal communities

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ABSTRACT

A meta-analysis approach was used to assess the effect of dredging induced changes in sediment composition, under different conditions of natural physical disturbance, for the structure and function of marine benthic macrofaunal communities. Results showed the sensitivity of macrofaunal communities increased as both the proportion of gravel increased and the level of natural physical disturbance decreased. These findings may be explained by the close association of certain taxa with the gravel fraction, and the influence of natural physical disturbance which, as it increases, tends to restrict the colonisation by these species. We conclude that maintaining the gravel content of surface sediments after dredging and, where practicable, locating extraction sites in areas of higher natural disturbance will minimise the potential for long-term negative impacts on the macrofauna.

1. Introduction

Marine aggregate dredging has the potential to lead to changes in the composition of seabed sediment habitats (Sarda et al., 2000; van Dalfsen et al., 2000). These changes can occur in four ways. Firstly, as a result of sediment screening, whereby unwanted sediment fractions (usually sands) are returned to the seabed (Hitchcock and Drucker, 1996; Newell et al., 1998, 2004). Secondly, as a result of the 'infilling' of dredge depressions and furrows with fine sediments (Dickson and Lee, 1972; Kenny et al., 1998; Boyd et al., 2002). Thirdly, as a result of 'overspill', where fine sediments, in suspension, are lost through chutes in the side of the dredger hold as the cargo is loaded (Newell et al., 1998). Fourthly, as a result of the exposure of underlying sediments which are different in nature to the original substrata (Kenny and Rees, 1996; Cooper et al., 2007). Material rejected by screening and overspill may also accumulate outside the boundaries of the extraction site, depending on local hydrodynamic conditions (Poiner and Kennedy, 1984; Hitchcock and Drucker, 1996; Newell et al., 2004; Cooper et al., 2006).

Changes in sediment composition can have implications for resident and recolonising fauna, resulting in the establishment of a faunal community that differs from the assemblage present before the dredging (Desprez, 2000; Boyd et al., 2002, 2005; Barrio Froján et al., 2011). Recognising the potential for such changes, Government policy (ODPM, 2002) requires developers to leave the seabed in a similar physical condition to that present before dredging. This measure is designed to enhance the possibility of, and rate at which, the seabed recovers physically and biologically to its pre-dredged condition. Whilst this policy has a clear scientific justification and is consistent with the principle of sustainable development, two important questions arise. Firstly, how can we decide what does, and does not, constitute an acceptable 'similar' physical condition? Secondly, how important is the preservation of sediment granulometry for faunal recovery in different localities? This question arises given the results from Kenny et al. (1991) and Rees et al. (1999), working at dredge sites off the east coast of the UK. They identified a combination of tidally induced sediment mobility and the abrasive effects of sand in suspension as important factors influencing benthic communities, and not simply sediment granulometry. The importance of sediment stability in controlling community structure is also highlighted in Newell et al. (1998). In addition, Seiderer and Newell (1999), Newell et al. (2001) and Cooper et al. (2007) all reported the a lack of a close correspondence between the distribution of different sediment types and benthic communities observed in the vicinity of marine aggregate dredging sites in areas of high natural disturbance. For this reason, Seiderer and Newell (1999) suggest that a return of sediment composition may not always be a pre-requisite for faunal recovery.

Improving our understanding of the relationship between sediment granulometry and the structure and function of macrofaunal communities is important for the management of marine

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aggregate dredging. For example, it will help identify those locations where it is more and less important to try to preserve sediment particle size composition, and hence where sediment screening should and perhaps should not be permitted. It will also help to determine whether there is a rational scientific justification for the active restoration of sediment particle size composition at sites of former marine aggregate dredging (e.g. Cooper et al., 2011a). The aim of this study was to determine to what extent changes in sediment composition, as a resulting of marine aggregate dredging, matter for the structure and function of benthic communities.

2. Methods

2.1. Data

A dataset comprising 368 samples of macrofauna and sediment particle size, from 12 individual surveys (11 sites) conducted between 2000 and 2008, was compiled for the purposes of this study. Each constituent dataset was originally collected to investigate the local impacts of marine aggregate dredging on macrofauna and sediments. All surveys included replicate samples taken from both a Reference site and one or more Treatment sites (Table 1). Reference sites were selected, typically using acoustic data, to be representative of the likely pre-dredge condition. All samples were acquired during the same survey using a 0.1 m^2 Hamon grab. Sub-samples were acquired for sediment particle size, with the remaining material processed for macrofauna over a 1 mm mesh sieve. Fauna were identified to the lowest level possible, usually species.

The extraction sites used in this study included actively dredged sites (Areas: 106, 408, 430, 447, 351, 122/3), and sites where dredging had ceased (Areas: 305, 222, X, Y, Owers). For the operational sites, only samples taken from Treatment sites outside the active dredge zones were included in the analyses. Samples taken within active dredge zones were not included in subsequent analyses as dredging was likely to be the overriding influence on assemblage composition. All extraction sites included in this study have been subject to commercial dredging, typically over many years, although the intensity of dredging, tonnages extracted and dredging techniques employed vary between sites. It should be noted that two datasets from Area 222 are included in the study. Both datasets, one from 2004 and the other from 2007, relate to different stages in a process of physical and biological recovery at this site.

To account for differences in the reporting of sediment particle size, these data were reduced to percentage fractions for gravel, coarse sand, medium sand, fine sand and mud. In addition, the species list was subjected to a process of rationalisation to account for different levels of taxonomic resolution between surveys. Biomass data were available for all surveys, with the exception of Area 305.

2.2. Site classification

In a Geographic Information System, maps of surface sediment distribution and natural physical disturbance were overlaid in order to produce maps showing the distribution and disturbance class of sand and gravel dominated sediments (Fig. 1). Both input layers were taken from Eggleton et al. (2011), although, for the purposes of the current study, the disturbance map was simplified into high, medium and low disturbance. The original seven disturbance categories were derived from a cluster analysis of georeferenced data, based on the following variables: mean annual chlorophyll concentration, mean annual suspended particulate matter, frequency of sediment reworking and sediment composition. The frequency of reworking is the number of days per year that the bed is disturbed as a result of shear stress induced by waves and tidal currents.

As the aggregates industry does not target muds these sediments were not considered further. The output maps identified six sediment classes. These included high, medium and low disturbance sands, and high, medium and low disturbance gravels.

Each of the 11 sites selected for this study were then overlaid on the output maps, allowing each site to be classified according to the dominant sediment type and the level of natural physical disturbance (Fig. 2). Where site specific sediment data (see Table 1) contradicted the broadscale information, then the site data were used in the classification. Area 408 was classified as 'Gravel_low disturbance', despite the sediment samples suggesting a dominance by sand, due to evidence of a gravel armouring reported in Cooper et al. (2005).

2.3. Assessment of benthic structure and function

The structure of macrofaunal communities was assessed using the species abundance data. The function of macrofaunal communities was assessed using four approaches: Somatic Production (Sp), Biological Traits Analysis (BTA), Infaunal Trophic Index (ITI), and the Rao coefficient (see Cooper et al., 2008 for detailed methodologies, and descriptions of the behaviour of each metric in response to marine aggregate dredging). Each of these techniques resulted in the formation of a separate multivariate dataset which was subjected to the same analyses as the species abundance data.

2.4. Data analysis

A total of six multivariate matrices were produced. These included one each for sediment composition and faunal structure, and four for faunal function (Table 2).

2.4.1. Univariate

The total number of species (S), abundance (N) and somatic production (Sp) were calculated for all samples. Differences in the mean value of each measure were compared between the different sediment/disturbance classes using graphical techniques.

2.4.2. Multivariate

Multivariate analyses were performed using PRIMER 6® (Clarke and Warwick, 1994). ANOSIM tests were used in order to determine the extent of any difference between each Treatment/Reference site pairing identified in Table 1, based on the six multivariate datasets listed in Table 2. A mean of the four functional ANOSIM *R* values was produced to represent faunal function in subsequent analyses. The *R* statistic from this test provides a measure of the similarity between two or more groups of samples. An *R* value of 0 indicates no difference between the groups, whilst an *R* value of 1 indicates a complete difference.

Based on each Treatment/Reference comparison, we then assessed the apparent sensitivity of faunal assemblages to changes in sediment particle size composition, using a weighted sensitivity index (*wSI*). Values of the index were calculated for both structural and functional sensitivity using the equations below.

$$wSI = \frac{R(\text{structure})^2}{R(psa)} \text{ and } wSI = \frac{R(\text{function})^2}{R(psa)}$$

The index is simply the ratio of the squared ANOSIM *R* value for faunal structure or faunal function, to the ANOSIM *R* value for sediment particle size data. Squaring the faunal *R* value increases the

Table 1
Details of the individual datasets used in this study, including: (i) survey information, (ii) site classification, according to both the dominant sediment composition of surface sediments, and the level of natural physical disturbance, (iii) Treatment and Reference sites available for comparison (number of replicates given in parenthesis) and (iv) sediment particle size composition of Reference site samples (based on a SIMPER analysis).

Survey					Site classification	Comparisons		Reference sample particle size composition		
Region	Site	Year (study reference)	Years since last dredged*	No. of samples	Sediment_Disturbance	Treatment(s)	Reference	% Gravel	% Sand	% Mud
Humber	Area 408	2004 (Cooper et al., 2005)	5	30	Gravel_Low disturbance	H (10) L (10)	R (10)	46	51	3
	Area 106	2003 (Newell et al., 2004)	0	56	Sand_Medium disturbance	High Intensity North (16) North 100 m (4) North 250 m (4) North 500 m (4) North 750 m (4) North 1250 m (4)	Control (20)	49	50	1
East Coast	Area 430	2003 (Newell et al., 2004)	0	78	Sand_Medium disturbance	North East Block (12) High Intensity North (10) North 100 m (4) North 250 m (4) North 500 m (4) North 750 m (4) North 1250 m (4) North 1750 m (4) North 2250 m (4) North 3250 m (4) North 4250 m (4)	Control (20)	40	58	2
	Area 305	2000 (Cefas, unpublished)	3	15	Sand_High disturbance	H (5) L (5)	R (5)	1	99	0
Thames	Area 222	2004 (Cooper et al., 2005)	7	30	Gravel_Medium disturbance	H (10) L (10)	R (10)	49	26	25
	Area 222	2007 (Wan Hussin et al., in press)	10	30	Gravel_Medium disturbance	H (10) L (10)	R (10)	47	33	20
	Area 447	2008 (Pearce et al., 2011)	0	22	Gravel_High disturbance	250 m (5) 650 m (6) 1000 m (6)	Control (5)	60	35	5
East English channel	Area X	2004 (Cooper et al., 2007)	2 (H) 7 (L)	30	Gravel_High disturbance	H (10) L (10)	R (10)	55	41	4
	Area Y	2004 (Cooper et al., 2005)	7	30	Gravel_High disturbance	H (10) L (10)	R (10)	55	41	4
South Coast	Owers	2000 (Cefas, unpublished)	1	15	Gravel_Low disturbance	H (5) L (5)	R (5)	64	33	3
	Area 351	2000 (Boyd and Rees, 2003)	0	16	Gravel_Low disturbance	C500 (4) C1000 (4) C2000 (4)	C5000 (4)	91	9	1
	Area 122/3	2000 (Boyd and Rees, 2003)	0	16	Gravel_Medium disturbance	D500 (4) D1000 (4) D2000 (4)	D5000 (4)	63	36	1
		Total		368						

* At the time of survey.

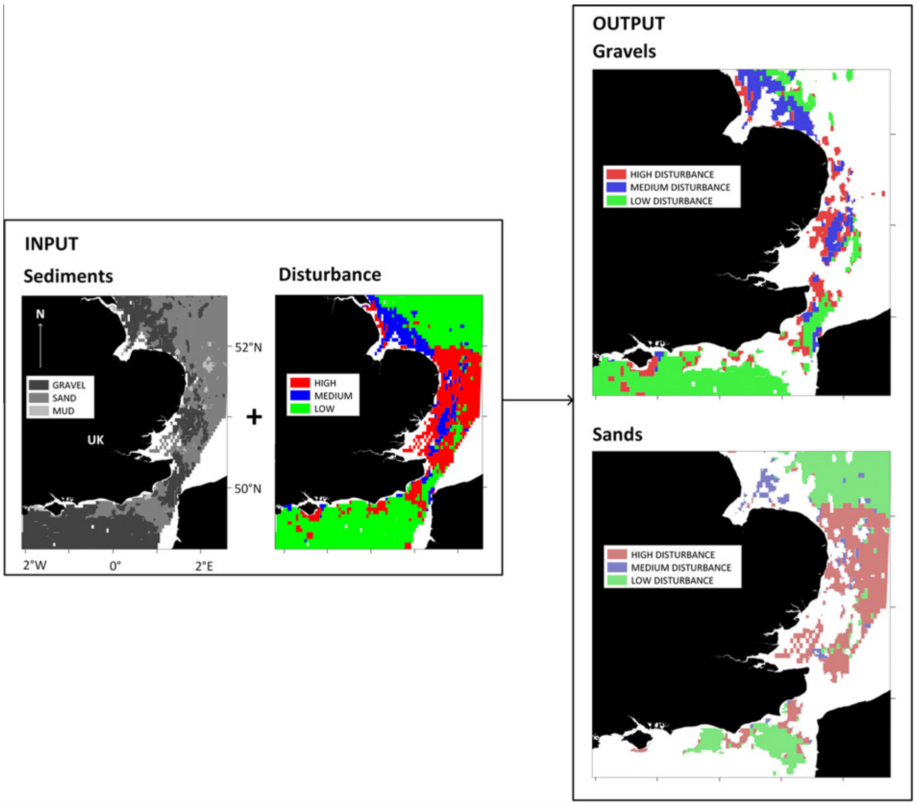

Fig. 1. Schematic diagram illustrating how input maps of (i) surface sediment distribution and (ii) natural physical disturbance were combined, in a GIS, to produce maps showing the distribution and disturbance class of sand and gravel dominated sediments. Both input maps were adapted from Eggleton et al. (2011).

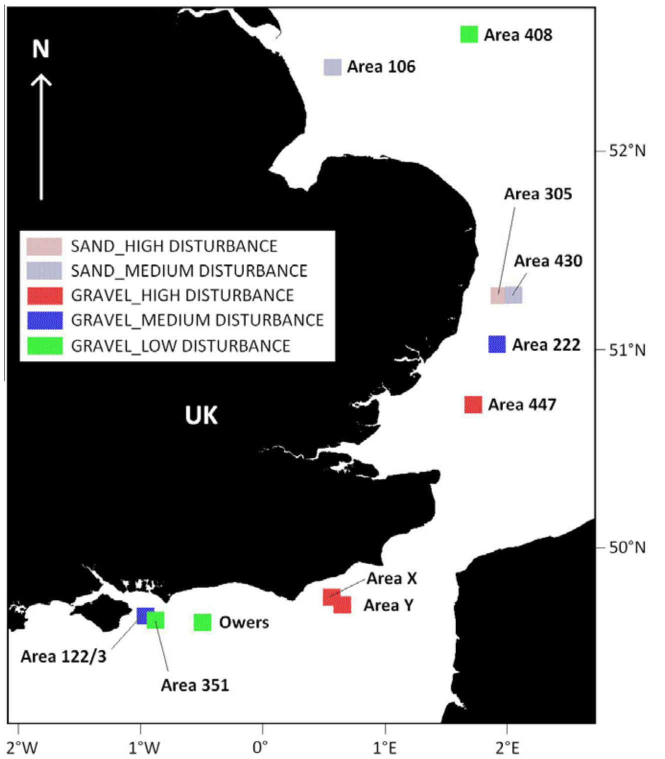

Fig. 2. Location and classification (sediment/disturbance) of study sites, based on the output maps in Fig. 1.

Table 2
Multivariate datasets for sediment, faunal structure and faunal function.

Data type	Dataset	Variables
Sediment	1. Particle size	% Major sediment classes
Faunal structure	2. Species abundance	Abundance values (species by station matrix)
Faunal function	3. BTA	Value of trait expression (trait by station matrix)
	4. ITI	Value of expression (trophic group by station matrix)
	5. Somatic production	Production values (family by station matrix)
	6. Rao	Rao coefficient values (trait by station matrix)

value of the index for larger faunal differences. This was considered important as there is likely to be greater concern over larger faunal differences.

For each sediment group, Values of *wSI* were plotted for each sediment/disturbance class, with groups arranged in order of increasing natural physical disturbance. The statistical significance of the resulting trend was then assessed using the non-parametric Mann–Kendall statistic (Mann, 1945; Kendall, 1975). We used the statistical programming environment *R* (R Development Core Team, 2010) to carry out the analyses. The Mann–Kendall statistic works by looking at each *wSI* value and counting the number of data points in higher groups that are greater than it (assigned +1) and also the number that are less than it (assigned −1). These values are summed over all *wSI* values to create a statistic, *M*.

The null distribution of *M* (i.e. assuming no trend) was calculated by Monte-Carlo simulation as follows. The observations were re-ordered at random so that they were potentially assigned to different groups and the value of M calculated. This was repeated 10,000 times. These values of M under the null hypothesis were compared against the observed value of M to estimate the p-value (Manly, 2001).

Given that ANOSIM R values do not indicate the direction of change – they merely identify the magnitude of any difference between samples from the Treatment and Reference sites – it is important to gain further insight into the nature of differences between Treatment and Reference sites. This was achieved by examining the percentage difference between sites in terms of the number of species, abundance, somatic production and the different sediment fractions. The average sediment composition of each site was determined using SIMPER analysis.

In order to explore differences in the nature of faunal communities from each sediment/disturbance class, a Principle Components Analysis (PCA) ordination, based on untransformed sediment particle size data (percentage contribution of gravel, coarse sand, medium sand, fine sand, mud) was produced. As variables were all in the same format it was not necessary to normalise the data. A SIMPER analysis, based on square-root transformed species abundance data, was used to identify the characterising species from both gravel and sand dominated samples. Bubble plots, overlaid on the PCA plot, were used to explore relationships between the abundance of individual species and sediment composition.

3. Results

3.1. Univariate

Despite the high variability, there was evidence for a general decline in the mean numbers of species, abundance and total somatic production as the level of natural physical disturbance increased (Fig. 3). In addition, within a disturbance class, mean values of all univariate measures were higher in gravel dominated samples compared with those dominated by sands, with the exception of total somatic production in areas of medium natural disturbance.

3.2. Multivariate

For both structural and functional *wSI*, there was evidence of a negative trend with increasing natural physical disturbance (M (structure) = 279; p-value = 0.001; M (function) = 205; p-value = 0.01) (Fig. 4). Having established differences between groups, we make the following observations. Firstly, that within a broad sediment group (sands or gravels), the sensitivity of faunal communities to changes in sediment particle size composition decreases as the level of natural physical disturbance increases. Secondly, that within disturbance groups (medium or low), assemblages found on gravel are more sensitive to changes in sediment composition than those on sand. Lastly, the converging of structural and functional *wSI* values suggests that assemblages may become less resistant to functional change as the level of natural physical disturbance increases.

On the whole, changes were considered to be detrimental. Of the thirty-seven Treatment/Reference comparisons, 76% resulted in a reduction in the number of species, 68% in a reduction in the number of individuals, and 69% in a loss of production (Fig. 5). Generally, faunal changes were associated with an increase in the proportion of sand and a decrease in the proportion of gravel (Fig. 5).

In addition, the relationship between the sediment particle size data of individual samples, and relative disturbance classes (see Fig. 1) was examined by PCA (Fig. 6a). Samples from low disturbance areas were dominated by gravel sediments, whilst samples

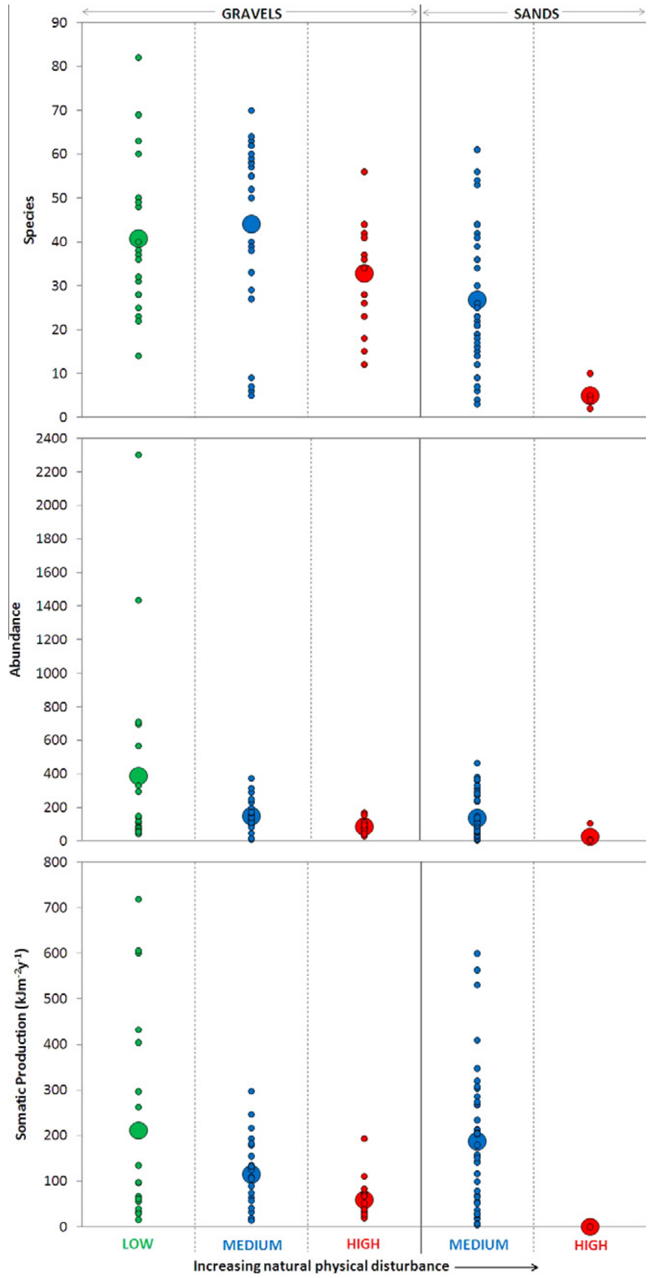

Fig. 3. Numbers of species, abundance and total somatic production in the reference samples belonging to each sediment/disturbance class. Sediment/disturbance classes are arranged in order of increasing natural physical disturbance. Mean values are shown by the large symbols.

from medium and high disturbance areas were quite evenly distributed across the sediment spectrum. Gravel sediments were generally associated with higher numbers of species, individuals and greater total somatic production (Fig. 6 b–d).

The abundance of individual characterising species, identified using the SIMER routine, was overlaid on the same PCA plot (Fig. 7). This suggests that many of the characteristic species in gravel dominated sediments (e.g. *Pomatoceros lamarcki*, *Balanus crenatus*, *Caulleriella alata*) favour coarse sediments (Fig. 7a). In contrast, many of the characteristic species found in sand dominated sediments (e.g. *Ophelia borealis*, *Glycera alba*, *Nephtys caeca*) can also be found in the sand component of gravel dominated sediments (Fig. 7b). For certain species common to both sand and

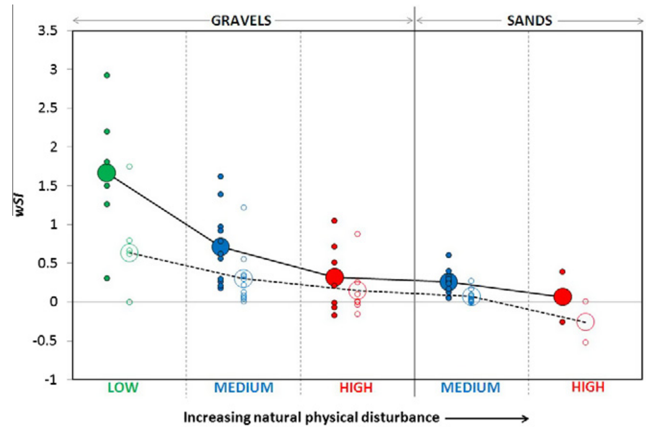

Fig. 4. Weighted sensitivity index (*wSI*) values for different sediment/disturbance classes, arranged in order of increasing natural physical disturbance. The structural *wSI* is shown by solid symbols, the functional *wSI* by open symbols. Mean values are shown by large symbols and joined by solid black line (structure) and dashed black line (function).

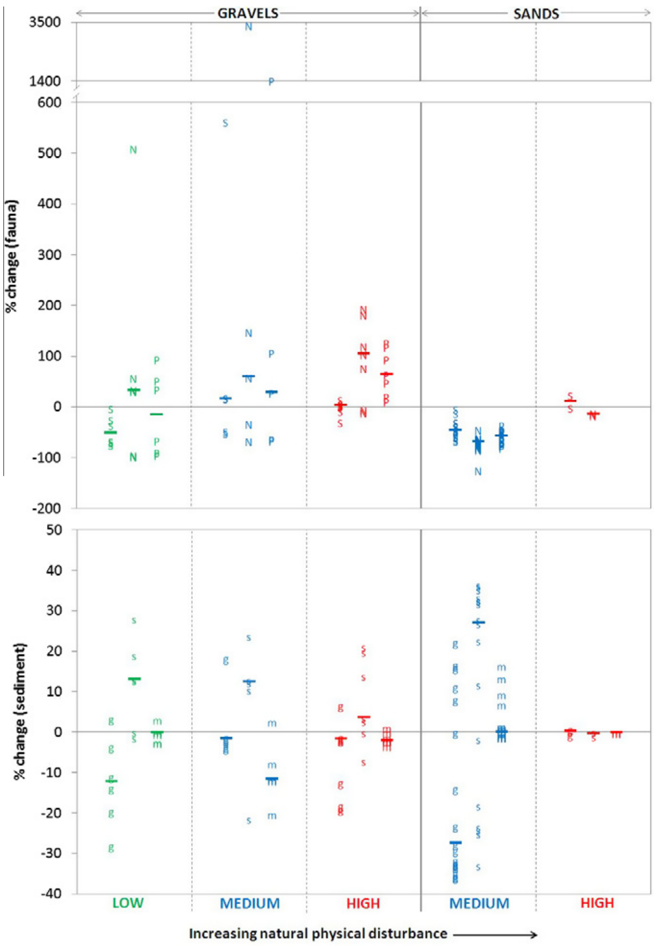

Fig. 5. Percentage change for (a) the number of species (S), abundance (N) and total somatic production (P). (b) The major sediment classes (g - gravel, s - sand, m - mud) at the Treatment site, relative to the Reference site (see Table 1 for sediment particle size composition of Reference site samples). Median values are shown by horizontal bars.

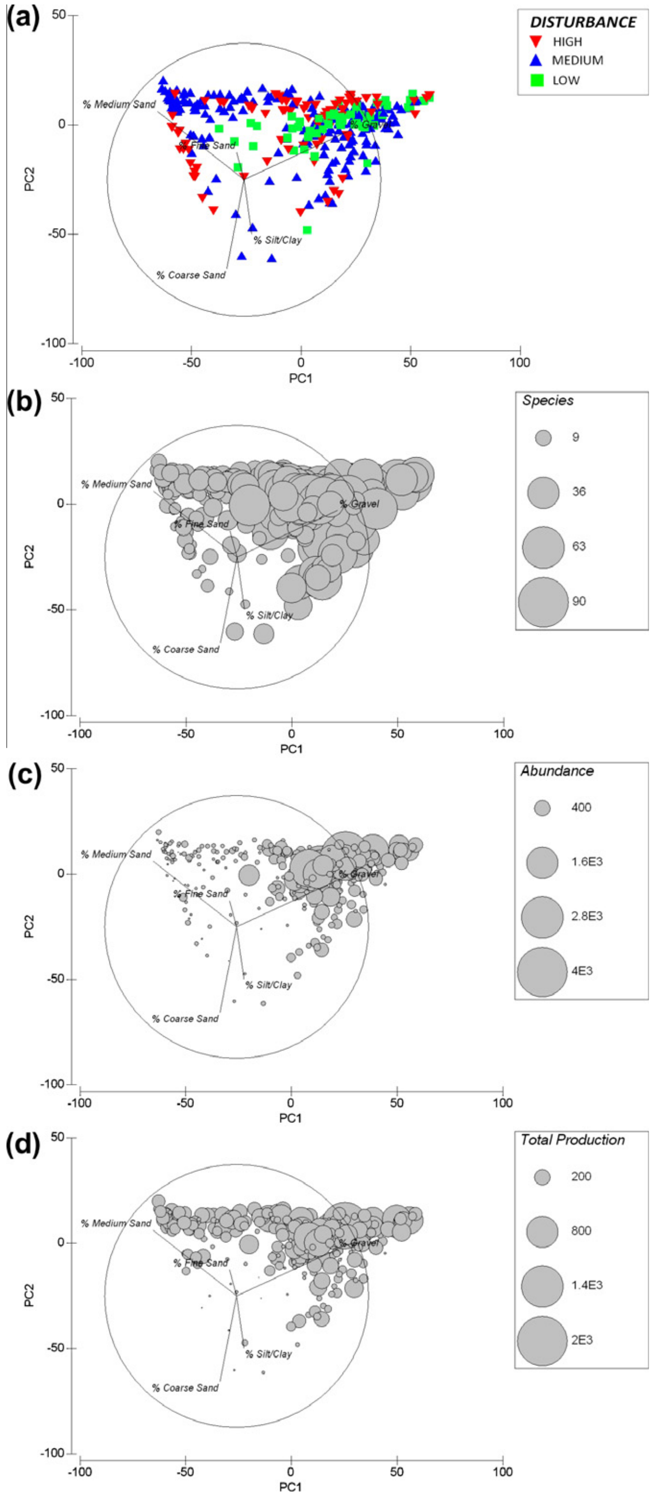

Fig. 6. (a) Principle Components Analysis (PCA) plot based on (a) sediment particle size data (samples coloured according to the level of natural physical disturbance), and superimposed bubble plots for (b) numbers of species (per 0.1 m^2), (c) abundance (per 0.1 m^2) and (d) total somatic production (kJ m^{-2} yr^{-1}).

gravel dominated sediments (e.g. *Lumbrineris gracilis*, *Aonides paucibranchiata*, *Sabellaria spinulosa*), the presence of gravel appears to be associated with an enhancement in their abundance (Fig. 7c).

4. Discussion

The purpose of this study was to investigate the significance of changes in sediment composition, as a result of marine aggregate dredging, for the structure and function of benthic macrofaunal

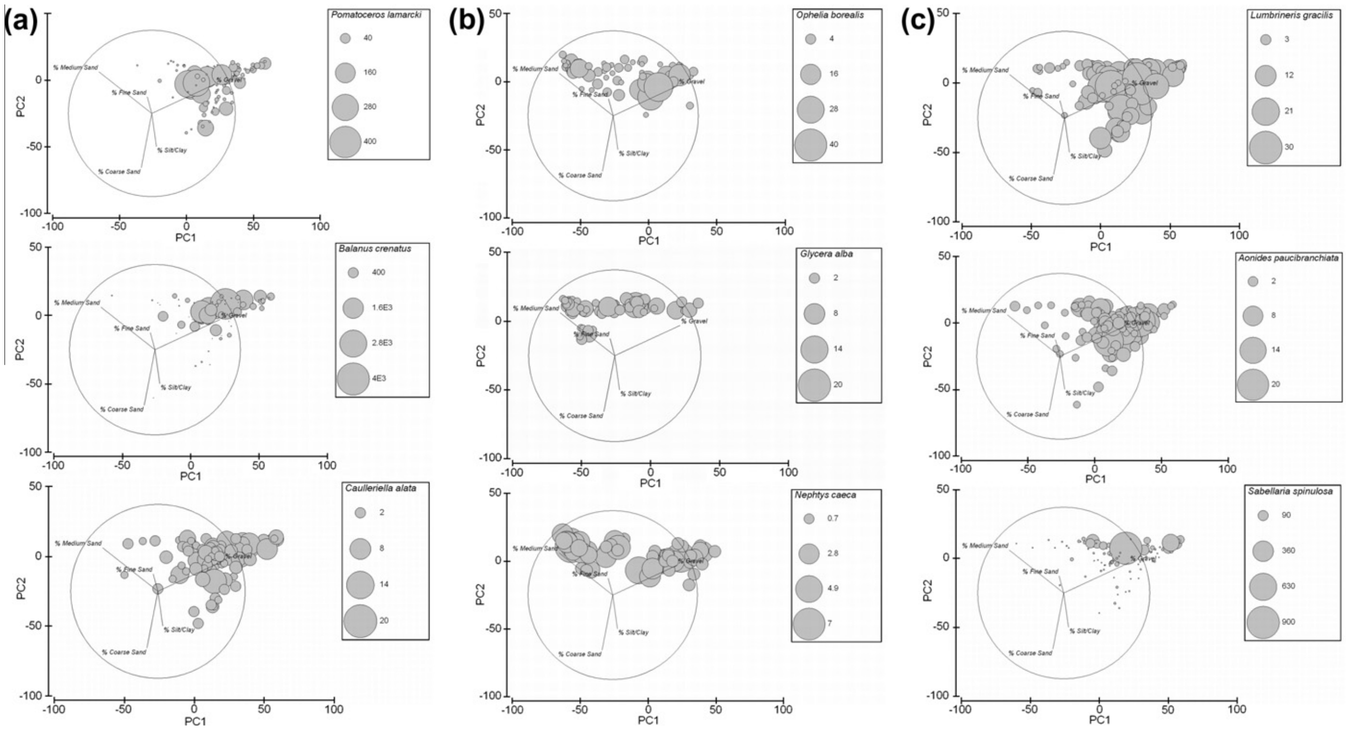

Fig. 7. Principle Components Analysis (PCA) plot – based on sediment particle size data – with superimposed bubble plots for: (a) abundance (per 0.1 m^2) of main characterising species found in gravel dominated sediments, (b) abundance (per 0.1 m^2) of main characterising species found in sand dominated sediments, and (c) abundance (per 0.1 m^2) of characterising species common to both gravel and sand dominated sediments.

communities. Data indicate that, within very broad terms, communities decline in species richness, abundance and productivity as (i) the level of natural physical disturbance increases and (ii) the proportion of gravel in samples decreases. This conclusion is in agreement with a study undertaken by Bolam et al. (2010) describing the productivity and diversity of UK shelf macrobenthic assemblages. However, faunal communities in areas of low natural physical disturbance, and with increased gravel content appear most sensitive to changes in sediment particle size composition. In contrast, faunal assemblages in areas of high natural physical disturbance, and with less gravel appear less sensitive to changes in sediment particle size composition. This observation accords with the findings of Seiderer and Newell (1999), Newell et al. (2001) and Cooper et al. (2007), who reported a lack of correspondence between community composition of the benthos and static particle size distribution in unconsolidated sand and gravel deposits at Area 452 (Thames), off Folkestone (eastern English Channel) and Cross Sands (east coast), respectively. All of these areas sit within zones of high natural physical disturbance as identified by this study. The differing sensitivity of faunal communities to changes in sediment particle size composition helps explain the different physical and biological recovery times following marine aggregate dredging reported in Foden et al. (2009).

Our results suggest the presence of gravel may have an important role in explaining the negative correlation between the sensitivity of faunal communities to changes in sediment composition and the level of natural physical disturbance. Many of the species characterising gravel dominated sediments were only found in association with this sediment fraction. Therefore, a loss of gravel, the most common sediment change observed in this study, can lead to a reduction in the abundance of these species. The effect of a loss in gravel is likely to be greater in areas of low natural physical disturbance where these 'gravely fauna' are most developed, and smaller in areas of high natural physical disturbance, where the effect of sand in suspension serves to limit settlement by such species (Kenny et al., 1991; Rees et al., 1999). In contrast to these gravely fauna, many of the characterising species from sand dominated sediments were equally likely to be found in gravel dominated sediments. As such, changes in sediment composition in these areas are likely to have a reduced impact on the overall faunal assemblage. However, although the impacts are reduced, the presence of gravel within sediments can be associated with an enhancement in abundance of a number of species commonly associated with sands and therefore some impacts are possible even in these more sand dominated habitats.

For these reasons, the policy (ODPM, 2002) which requires operators of marine aggregate extraction sites to leave sediments in a 'similar' physical condition to those present prior to dredging appears sensible. However, this study suggests the requirement will be more or less important depending on the nature of the local environmental conditions. In addition, the approach taken in this study has shown that it may be possible to work towards the setting of specific and measurable limits for changes in sediment composition. This is necessary if licence conditions regarding changes in sediment composition are to be enforceable.

Clearly some caution must be exercised in drawing firm conclusions about the differences between sediment/disturbance classes due to the limited and unequal number of data points. Never-theless, we cautiously identify a number of theoretical and practical implications resulting from our findings. Most obvious is that it may be preferential, where a choice exists, to site new aggregate dredging licences in areas of high natural disturbance. In addition, efforts should be made, so far as is practicable, to seek to maintain a similar quantity of gravel in surface sediments after dredging, particularly in areas of low natural physical disturbance (e.g. through limitations on sediment screening in areas where such material may persist). This is important given the role of gravel in providing a surface for attachment of some species, and in stabilising sands and finer sediments. Such measures will help reduce the likelihood of permanent changes in faunal community composition.

Where changes in sediment composition do arise, despite appropriate licence conditions being in place and adhered to, this raises question of what is an appropriate management response. The currently prevailing view is to accept that a degree of impact may be inevitable, but that changes are generally acceptable when balanced against the societal benefits associated with extraction of marine aggregates. The relatively small size of impacted areas strengthens this view. However, there remains the potential for cumulative and in-combination effects. In response to this, recent work shown that it is possible to, at least in part, restore sediment composition following marine aggregate dredging (Collins and Mallinson, 2006; Cooper et al., 2011a). Whilst the present study strengthens the case for restoration, further work is required to determine whether the costs of such intervention can be justified, both scientifically and in terms of economics (Cooper et al., 2011b).

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Assessing the recovery of functional diversity after sustained sediment screening at an aggregate dredging site in the North Sea

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ABSTRACT

The effects of dredging the seabed for aggregate on benthic functional diversity were assessed using a suite of suitable indices on a recovering macrofaunal assemblage. Recovery was assessed as the return of a dredged assemblage to a state found in neighbouring undisturbed (reference) sites. *In situ* sediment screening was permitted during dredging operations; a difference in the sedimentary profile of the seabed between dredged and undisturbed reference sites was also observed. At sites of relatively high and low dredging intensity the sediment appeared more homogenous than reference sites after the selective removal of the coarser component. Initial assessment of the macrofaunal assemblage using univariate analytical techniques suggested a recovery of functional diversity at the low dredging intensity site after two years (according to the Infaunal Trophic Index, Taxonomic Distinctness index and Rao's Quadratic Entropy coefficient). However, multivariate analyses of the same data and of all indices except Taxonomic Distinctness indicated that assemblages at both high and low dredging intensity sites remained statistically indistinguishable from each other yet markedly different to the assemblage present in the reference area during the four-year study. The study concluded that recovery of functional diversity to a level found in a neighbouring undredged habitat had not occurred at either dredged site five years after the cessation of dredging. It is thought that the damage by dredging to functional diversity and to the capacity of the macrofaunal assemblage to recover is immediate and not so dependent on dredging intensity. The cumulative and wider ranging effects of sediment screening cannot be ignored or dismissed as a contributing factor to the similarities observed. The wider significance of these findings on the regulation of dredging activities is discussed.

1. Introduction

Research into the effects of aggregate dredging on the marine environment has been growing steadily since the practice of sourcing aggregate from the seabed became commonplace in the UK in the 1960s (The Crown Estate, 2009). The ultimate purpose of such research is to understand the consequences of aggregate dredging in order to inform the regulatory process through which conditional extraction licences are granted. Research has so far covered some important issues related to sediment movement (e.g., Brampton, 1993; Burningham and French, 2008, 2009a,b), coastal impacts (e.g., Balson et al., 2007; Cooper et al., 2008a) and environmental impacts (e.g., Boyd et al., 2004; Cooper et al., 2005; Robinson et al., 2005; Barrio Froján et al., 2008). Of the latter research theme, most attention has been focused on assessing the rate and extent of recovery of the resident benthic faunal assemblages using standard diversity metrics. Findings to date have been instrumental in the setting of current licence conditions that aim to mitigate the physically disruptive and ecologically disturbing practice of dredging for aggregate. The research presented in this communication aims to improve our understanding of the recovery process by (a) investigating how an ecosystem's functional diversity has been affected by dredging, and (b) measuring the rate at which a dredged ecosystem returns to a functional condition similar to that of a nearby undisturbed area. Any effects of dredging on the long-term functional capacity of an ecosystem should be of crucial importance in the setting of future licence conditions to ensure that no lasting damage is inflicted on the environment and its ability to perform its various functions.

Many variables affect biological recovery after disturbance by dredging. These range from those inflicted deliberately through the

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physical manipulation of the environment – such as the extent, intensity, duration and frequency of dredging and the resulting sedimentary profile of the affected seabed – to natural fluctuations in population dynamics, local hydrographic and climatic conditions, and the sedimentary composition of the surrounding area (Newell et al., 1998; Dernie et al., 2003; Boyd et al., 2004; Cooper et al., 2005). The ability to interrogate the influence of any or all of these variables on the recolonising faunal assemblage remains a challenge. To address this challenge, traditional methods of assessing change in biological assemblages have used a number of metrics, such as counting organisms and species, or the calculation of standard diversity indices, and comparing these under various sets of conditions through time. However, the use of these relatively simplistic metrics can often reflect an equally simplistic view of the changes that have taken place. For instance, a recovering assemblage may quickly regain the same abundance of organisms or diversity of species that it had prior to being disturbed, but the new assemblage may not be able to perform the same range of ecological functions as that which it has replaced. Equally, the converse can also occur, where an assemblage may not regain its previous level of richness or diversity but can still perform the same range of ecological functions (Peterson et al., 1998; Cardinale et al., 2000, 2002; Palmer et al., 2008). Either way, traditional metrics and methodologies do not have the capacity to detect or reflect such subtle and complex interactions.

A suite of metrics and methodologies have recently been developed to detect changes in various aspects of ecosystem function. This is no easy feat, as ecosystem function is a multifaceted concept that incorporates the interdependence of organisms with each other and with the environment, encompassing energy flow, mineral, nutrient and water cycling, habitat transformation, and social interaction and succession (de Groot, 1992; Schulze and Mooney, 1994; de Groot et al., 2002). Clearly, no index could capture all that complexity in a single figure. They do, however, each capture some element of ecosystem function and by using a number of different techniques, a weight of evidence can be amassed to support or refute any pattern that may be observed. Furthermore, many of the indices developed have been applied mostly to terrestrial datasets, with only a few attempts to use them in the marine environment (but see Bremner et al., 2003; Diaz et al., 2004). Cooper et al. (2008b) recently used a selection of such indices (including functional diversity indices) to examine how aggregate extraction activities designed not to alter the sediment profile of the seabed affect functional diversity and the ecosystem's capacity to recover after disturbance. In the present study the same indices are calculated under an alternative scenario, where aggregate extraction, through utilisation of a sediment screening process, changes the physical characteristics of the seabed. Benthic assemblages are closely related to the form of the sedimentary environment (Snelgrove and Butman, 1994), so changes in this environment can have implications both for their structure and function.

2. Methods

2.1. Study site and data acquisition

Area 408 is a licensed aggregate extraction area that lies 100 km east of the Humber Estuary, in the southern North Sea (Fig. 1). Between 1996 and 1999, active dredging and sediment screening for gravel took place within a sub-sector of the licensed area, known as Dredge Zone 2. Here, between 1 and 14 h of dredging took place each year within any 100×100 m box (The Crown Estate and BMAPA, 2005). Electronic Monitoring System data (EMS – a GPS device fitted to every dredger that records its position and activity every 30 seconds) was used to delimit two boxes (300×300 m each) representing areas of high and low dredging activity. Within each of these areas, 10 replicate sediment samples were collected using a mini Hamon grab (0.1 m^2) in May/June each year between 2001 and 2004. In addition, two sites outside Area 408 were sampled (5 replicates from each) at the same time and over the same period to provide a reference point. Over the four-year sampling period, a total of 120 samples were taken (10 replicates from the Reference, High and Low dredging intensity sites each year). Sediment samples were processed for macrofauna and particle size distribution (PSD) following the guidelines in Boyd (2002). Faunal identification was conducted to the lowest possible taxonomic level.

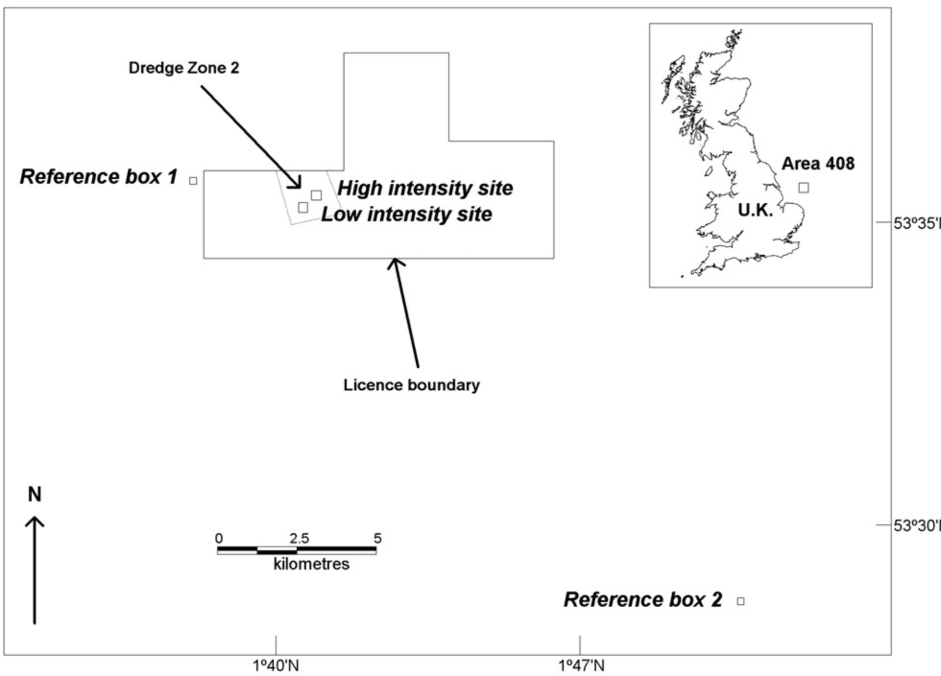

Fig. 1. Map of study area indicating the High and Low dredging intensity sites as well as the Reference sites within and around Dredge Zone 2 of Area 408.

2.2. Data analysis

The acquired biological dataset consisted of 312 taxa (hereon in referred to as species), of which 41 were colonial. Colonial species have been excluded from analyses requiring abundance information. Non-colonial species were represented by 12,972 individuals. Preliminary analyses involved the calculation of biomass (Ash Free Dry Weight in grams) and Hill's (1973) diversity and evenness indices (N1 and N2, respectively) for each sample. Each species was then scored according to how it represented a selected set of traits (see Table 2 in Cooper et al. (2008b) for the list of traits and the different categories within each). This information, together with an aggregation file (in which the taxonomic hierarchy of each species is tabulated) was used in the calculation of the different indices representing functional diversity. Chosen indices were: (1) the Infaunal Trophic Index (ITI – calculated using results from Trophic Group Analysis TGA), (2) Somatic Production (P_S), (3) Biological Traits Analysis (BTA), (4) Taxonomic Distinctness (TD), and (5) Rao's Quadratic Entropy Coefficient (Q). A summary of the rationale behind each index and references to the original publications detailing the methodology of their calculation can be found in Cooper et al. (2008b). Indices were calculated for each sample and samples grouped by treatment and/or year to calculate and compare the mean values of each index under different conditions. Calculations and statistical comparison of means were performed using one-way ANOVA in the Minitab 15 statistical software package.

Multivariate analyses were performed using the PRIMER 6 software package (Clarke and Gorley, 2006). Conventional multivariate analyses use a variable-by-sample matrix as a starting point, where the variables are usually species and their respective abundance or biomass values in each sample. In addition to these conventional analyses, this study used variable-by-sample matrices where variables were either trophic groups (as defined by TGA), or the scores of an index calculated for each different trait (e.g., the Rao's Q or BTA value for each of the selected traits); so, effectively, a multivariate resemblance measure was calculated from an index value-by-sample matrix. Resulting resemblance matrices were converted to dendrograms using group averaging in the CLUSTER routine, together with SIMPROF and ANOSIM tests where appropriate, to test for statistically significant differences between predefined groups of samples. SIMPER tests served to identify which variable contributed most to any observed differences in the data. Correlations between the resemblance matrix based on particle size distribution (PSD) data and those based on biotic variables were calculated using the RELATE routine. A funnel plot (TAXDTEST) was used to assess the degree of departure from expectation of the Taxonomic Distinctness values calculated for different treatments (i.e., Reference, High and Low dredging intensity). Lastly, the proportion of each sediment fraction in each sample was used to construct a Principal Component Analysis plot and conduct other relevant multivariate analyses.

3. Results

3.1. Physical effects

Each year, sediments at the Reference sites were clearly distinct from those at dredged sites (Fig. 2). The sediment fractions responsible for the observed differences appeared to be a greater proportion of gravel, fine sand and silt/clay at Reference sites than at dredged sites, and a greater proportion of coarse and medium sand at dredged sites than at Reference sites. High and Low dredging intensity sites appeared to be separated by virtue of mean sediment size, with the High intensity site having an overall greater mean particle size than the Low intensity site. Multivariate analysis of the different sediment fractions by sample (averaged by site each year) revealed that the sediment profile of each site was significantly different (Fig. 4a – sites/years joined by dotted lines are statistically indistinguishable from each other ($p > 0.05$)). This is supported by results from a two-way ANOSIM test on all samples testing for differences in sediment profile between sites and years; differences between years across all sites were not significant, whereas differences between sites across all years were significant (ANOSIM $R = 0.33, p < 0.01$). Pairwise tests between all three sites irrespective of year all revealed significant differences between sites, but differences between Reference and the other two sites were greater (ANOSIM $R > 0.41$, $p < 0.01$) than those between High and Low intensity sites (ANOSIM $R = 0.14, p < 0.05$). Correlations between the resemblance matrix based on PSD data and all other resemblance matrices based on biotic data (described below) were significant (Spearman's Rho ≥ 0.50, $p = 0.04$), indicating that patterns in sediment composition across the study site had some influence on the observed patterns in faunal assemblage composition.

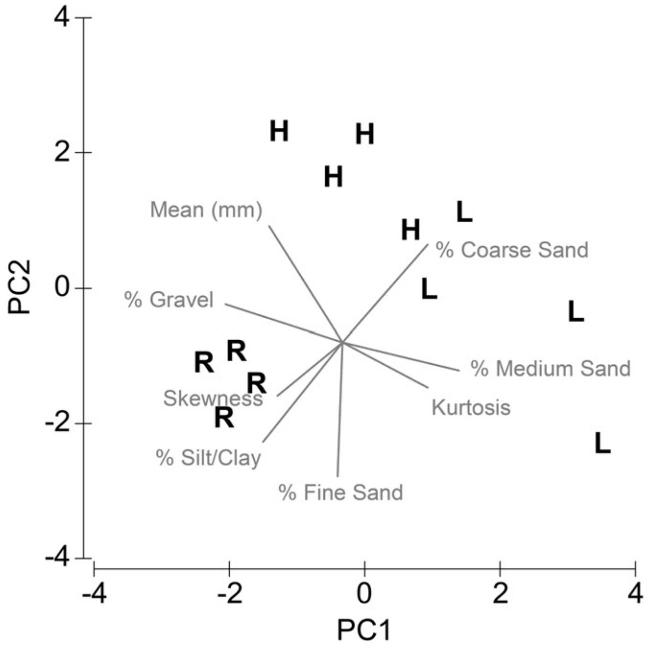

Fig. 2. Plot of Principal Components Analysis on normalised sediment metrics (covariant metrics removed). Each H, L and R symbol plotted represents a year between 2001 and 2004 at the High, Low and Reference sites, respectively.

3.2. Biological effects

3.2.1. Abundance, biomass and number of species

Mean macrofaunal abundance values were consistently higher (ANOVA $F > 53.3$, $p < 0.05$, d.f. $= 2$) at Reference sites than at dredged sites, but no difference in abundance was observed between High and Low dredging intensity sites (Fig. 3a). A surge in the number of individuals under Reference conditions during 2004 was caused by the sudden appearance of members of the barnacle species *Balanus crenatus* at the northernmost Reference site. Mean biomass and the mean number of species also followed the same pattern; however, the surge in *B. crenatus* did not appear to distort the biomass value recorded in 2004 at the Reference site (Fig. 3b and c). The assemblage at the Reference sites was statistically indistinguishable from one year to the next, in the same way that the assemblages at each of the dredged sites were also indistinguishable from each other over the years (Fig. 4b). Differences were apparent in all years between the assemblages at the Reference

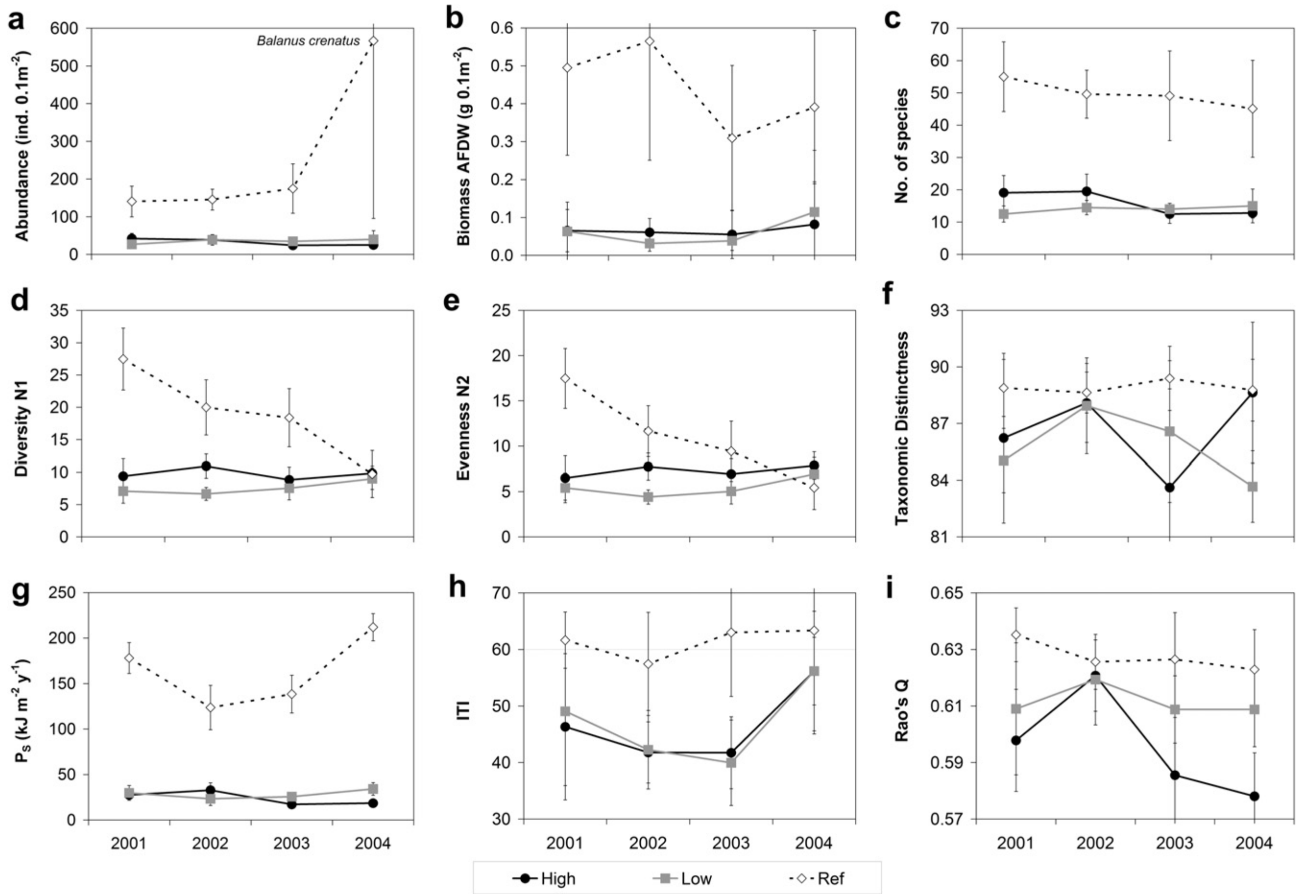

Fig. 3. Plots of mean index value (±95% Confidence Intervals) at each site (High, Low, Reference) over time. Indices/metrics plotted are: (a) abundance, (b) biomass, (c) number of species, (d) species diversity, (e) species evenness, (f) Taxonomic Distinctness, (g) production, (h) Infaunal Trophic Index scores and (i) Rao's Quadratic Entropy coefficient.

sites and those at the High and Low dredging intensity sites (Fig. 4b). A SIMPER test revealed that the assemblages at both Reference and dredged sites were 66.5% dissimilar. The species principally responsible for this dissimilarity – *B. crenatus*, *Urothoe elegans*, *Mysella bidentata*, *Urothoe marina* and *Phoronis* sp. (Table 1) – were more abundant at the Reference sites than at the dredged sites. Significant differences were evident between all three treatments, although the difference between both dredged sites was of a much lesser magnitude (ANOSIM R statistic = 0.34, $p < 0.01$) than that between each dredged site produced and the Reference sites (ANOSIM R statistics ≥ 0.75, $p < 0.01$). The same patterns were observed when biomass, species presence/absence and non-transformed abundance data were used. It would appear that full recovery of assemblage structure at dredged sites, as judged by the return of abundance, biomass and species number to reference conditions, has not yet taken place (Table 2).

3.2.2. Diversity and evenness

Diversity and evenness indices attempt to capture the variety and distribution of organisms among species (Washington, 1984). Mean species diversity (N1) and evenness (N2) were significantly higher at the Reference sites than at the dredged sites during all years except 2004 (Fig. 3d and e). There was no difference in either index between High and Low dredging intensity sites. In 2004, mean diversity and evenness values at the Reference sites dropped to levels similar to those at the dredged sites. The sudden appearance of *Balanus crenatus* at one of the Reference sites is likely to have influenced this result, since both indices rely on relative species abundance values for their calculation. As the diversity and evenness index values at the Reference sites decreased rather than the values of the dredged sites increasing, recovery of diversity and evenness at the High and Low dredging intensity sites cannot be said to have occurred (Table 2).

3.2.3. Taxonomic distinctness

Taxonomic Distinctness (TD) attempts to capture phylogenetic diversity; it is assumed that a more phylogenetically diverse assemblage accommodates a more diverse range of functional traits (Clarke and Warwick, 1998, 1999). Year by year, mean values of TD at the Reference sites remained constantly high, whilst at both the High and Low intensity dredging sites, mean TD values fluctuated considerably (Fig. 3f). However, only in 2004 were TD values at the Low intensity site significantly different to TD values at other sites. TD values at the High and Low intensity sites did not follow a similar trend throughout the four-year study period. In no year did mean TD values at any site depart significantly from expectation, as evidenced in the funnel plot in Fig. 5. Based on the TD index, the assemblages at both dredged sites are as taxonomically distinct – and by inference as phylogenetically and functionally diverse – as the assemblage at the Reference sites.

3.2.4. Somatic production

Somatic Production (P_S) is the quantity of matter/energy which is potentially available as food for the next trophic level (i.e., for natural predators) (Brey, 2001; Cusson and Bourget, 2005). P_S was considerably higher at the Reference sites than at both dredged sites throughout the study period (Fig. 3g). High and Low dredging intensity sites were indistinguishable in terms of their mean P_S

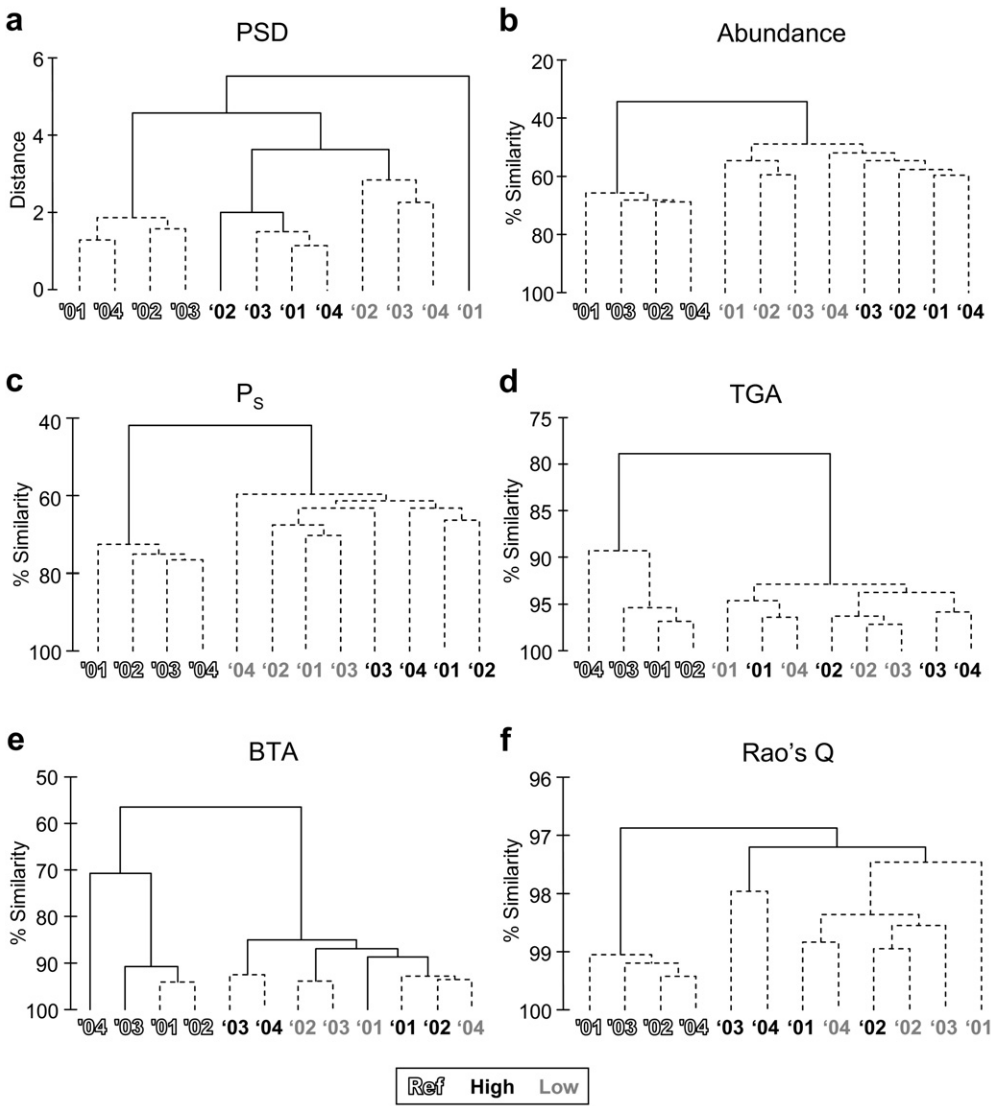

Fig. 4. Dendrograms representing the similarity between sediments and macrofaunal assemblages sampled each year at each of the sampling sites (High site represented by black years, Low site represented by grey years, Reference sites represented by hollow years) as measured by each analytical technique. Metrics analysed are (a) PSD data, (b) abundance, (c) production, (d) Trophic Group Analysis scores (used to calculate ITI), (e) Biological Trait Analysis scores and (f) Rao's Quadratic Entropy coefficient. Dotted lines illustrate the result from the SIMPROF routine, indicating no statistical difference amongst assemblages connected by them.

values. This pattern was also observed in multivariate analyses (Fig. 4c), where the P_S of the assemblage sampled each year at the Reference sites remained similar over time, yet differed from that of the High and Low dredging intensity sites (as indicated by the SIMPROF test). According to a SIMPER procedure, these two distinct groups were 59.4% dissimilar, and the productive output of the top 5 taxon groups responsible for this dissimilarity (Table 1) was greater at the Reference sites than at the dredged sites. There was no overall difference in P_S between years, but a significant difference between Reference sites and both dredged sites was detected (ANOSIM R statistics ≥ 0.74, $p < 0.01$). The difference between High and Low intensity sites, although significant, was of far lesser magnitude (ANOSIM R statistic $= 0.32$, $p < 0.01$) than between Reference and dredged sites. Based on results from analyses on Somatic Production, recovery of ecosystem function has yet to occur at the High and Low dredging intensity sites (Table 2).

3.2.5. Infaunal trophic index

The Infaunal Trophic Index (ITI) and the Trophic Group Analysis (TGA) from which it is calculated are based on organisms' feeding habits, which are thought to be one of the mechanisms underlying ecosystem function (Pearson and Rosenberg, 1978, 1987). Mean ITI values each year at the Reference sites were either above or not significantly different from the 60 score threshold indicative of an undisturbed environment (Fig. 3h). Mean ITI values at the High and Low dredging intensity sites were constantly below the 60 score threshold, indicating a modified/disturbed environment. However, both in 2001 and 2004, the ITI scores for both dredged sites were not significantly different to those of the Reference sites. Also, in 2004 the ITI score of both dredged sites was below but not significantly different from the 60 score threshold. Multivariate analysis of the trophic group data separated Reference sites from dredged sites, with the assemblages within those two distinct groups being indistinguishable over the years (Fig. 4d). Of the 4 pre-defined trophic groups, the difference in the number of organisms belonging to Trophic Group 1 (TG1 – suspension or detrital feeders) had the greatest influence in separating Reference from dredged sites (Table 1), followed by TG2 (surface or interface detritus feeders), TG3 (surface deposit feeders) and TG4 (sub-surface deposit feeders). Two two-way crossed ANOSIM tests were

Table 1
A list of the top 5 variables that contributed the most to the observed dissimilarity between different assemblages as defined by multivariate analyses (SIMPER test).

Metric/variable	Ref. Av. contrib.	High & low Av. contrib.	Av. diss.	% Contrib.	% Cum. contrib.
Abundance					
Balanus crenatus	1.66	0.69	0.93	1.40	1.40
Urothoe elegans	2.01	0.54	0.92	1.38	2.78
Mysella bidentata	1.60	0.15	0.91	1.36	4.14
Urothoe marina	1.85	0.44	0.88	1.33	5.46
Phoronis	1.28	0.00	0.81	1.21	6.68
Somatic production					
Upogebiidae	2.32	0.25	2.12	3.57	3.57
ACTINIARIA	1.43	0.16	1.33	2.23	5.80
Sabellidae	1.25	0.00	1.29	2.16	7.96
Sigalionidae	1.19	0.11	1.14	1.93	9.89
Scalibregmatidae	1.17	0.06	1.14	1.91	11.80
Trophic group analysis					
TG1	3.20	1.61	9.20	43.54	43.54
TG2	2.56	1.71	5.10	24.12	67.65
TG3	2.37	1.61	4.58	21.64	89.29
TG4	2.06	1.69	2.26	10.71	100.00
Biological traits analysis[a]					
F – suspension/filter	8.54	1.74	1.87	4.30	4.30
S – 0.5–1.0 cm	8.84	2.04	1.86	4.26	8.56
A – permanent attachment	9.73	2.64	1.85	4.25	12.81
A – no attachment	10.99	5.09	1.76	4.04	16.85
L – direct development	7.94	2.24	1.72	3.94	20.79
Rao's Q					
Larval type	0.63	0.56	0.77	24.56	24.56
Feeding habit	0.72	0.67	0.56	17.79	42.36
Size	0.76	0.72	0.48	15.43	57.79
Body form	0.72	0.68	0.42	13.27	71.06
Degree of attachment	0.37	0.39	0.34	10.98	82.04

[a] Trait key: F = feeding habit; S = size; A = adult life habit; L = larval type.

Table 2
Recovery times of the high and low dredging intensity sites according to each of the indices of ecosystem function tested. > and < before a year indicate whether recovery has occurred 'after' or 'before' that year, respectively.

Metric/index	Analysis technique	Year of recovery (number of years after dredging) Low intensity site	High intensity site
Abundance	Univariate	>2004 (>5)	>2004 (>5)
	Multivariate	>2004 (>5)	>2004 (>5)
Biomass	Univariate	>2004 (>5)	>2004 (>5)
	Multivariate	>2004 (>5)	>2004 (>5)
Number of species	Univariate	>2004 (>5)	>2004 (>5)
	Multivariate	>2004 (>5)	>2004 (>5)
Diversity (N1)	Univariate	>2004[a] (>5)	>2004[a] (>5)
Evenness (N2)	Univariate	>2004[a] (>5)	>2004[a] (>5)
Tax. distinctness	Univariate	≤2001 (≤2)	≤2001 (≤2)
	Multivariate	≤2001 (≤2)	≤2001 (≤2)
Somatic production	Univariate	>2004 (>5)	>2004 (>5)
	Multivariate	>2004 (>5)	>2004 (>5)
Infaunal trophic index	Univariate	2004[b] (5)	2004[b] (5)
	Multivariate	>2004 (>5)	>2004 (>5)
Biological traits analysis	Multivariate	>2004 (>5)	>2004 (>5)
Rao's Q	Univariate	≤2001 (≤2)	>2004 (>5)
	Multivariate	>2004 (>5)	>2004 (>5)

[a] Although no statistical difference was evident between reference and dredged sites, this was due to a perceived reduction in diversity and evenness at reference sites, not a return to naturally high diversity and evenness at dredged sites.
[b] Although in 2001 there was no statistical difference between the mean ITI scores at reference and dredged sites, the latter were still significantly below the 60 score threshold, therefore indicative of a disturbed environment.

performed on TGA data for each sample, the first to test for differences in assemblage trophic composition between treatments (Ref., High and Low) over the years and the second to test for differences between pre-defined categories of impact (i.e., samples with an ITI index between 60 and 100, indicative of an undisturbed environment; samples with an ITI index between 30 and 60, indicative of a modified environment; and samples with an ITI index value below 30, indicative of a degraded environment). In the first test, pairwise comparisons between years revealed that the whole assemblage did not change significantly over time (ANOSIM $R = 0.09$, $p < 0.01$); however, pairwise comparisons between sites revealed that the assemblage at the Reference sites was significantly different to those inhabiting the dredged sites (ANOSIM $R > 0.61$, $p < 0.01$), and no difference existed between assemblages at the High and Low dredging intensity sites (ANOSIM $R = 0.04$, $p = 0.67$). In the second ANOSIM test, there was a significant difference in assemblage trophic composition between 'undisturbed' and 'modified' samples (ANOSIM $R = 0.44$, $p < 0.01$), and between 'undisturbed' and 'degraded' samples (ANOSIM $R = 0.38$, $p < 0.01$). There was no difference between 'modified' and 'degraded' samples (ANOSIM $R = 0.04$, $p = 0.34$). In other words, the assemblage trophic composition in 'modified' and 'degraded' samples was the same. It would appear that assemblage trophic composition has not yet recovered fully at dredged sites (Table 2).

3.2.6. Biological traits analysis

Biological Traits Analysis (BTA) uses morphological, behavioural and life history characteristics of species in an assemblage to give an indication of its functional diversity (Bremner et al., 2003). Multivariate analysis of BTA scores (using square-root transformation) clearly separated the assemblage at the Reference sites from those at the High and Low dredging intensity sites (Fig. 4e). The dredged and Reference sites were 43.6% dissimilar, being separated mainly by differences in the occurrence of suspension/filter feeders, organisms of 0.5-1.0 cm length, permanently attached fauna, unattached fauna and those with direct development (Table 1). The assemblage sampled at both dredged sites was 85.2% similar, but also contained statistically distinct sub-assemblages occupying both sites over the four-year study period. The Reference sites appeared to show some

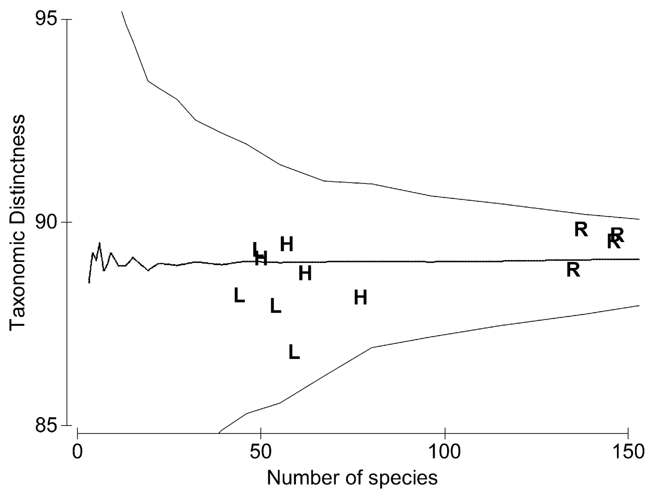

Fig. 5. Funnel plot of Taxonomic Distinctness against increasing numbers of species (centre line) with 95% Confidence Interval (funnel). Each H, L and R symbol plotted represents a year between 2001 and 2004 at the High, Low and Reference sites, respectively.

temporal change after 2002, though a lack of a similarly clear pattern at dredged sites from year to year suggests any change is limited to the Reference site and not a general phenomenon. ANOSIM tests revealed a significant difference between assemblages at Reference and dredged sites (ANOSIM $R > 0.71$, $p < 0.01$), but not between High and Low dredging intensity sites (ANOSIM $R = 0.18$, $p < 0.01$). Similar patterns were observed using untransformed and fourth-root transformed data (not shown), suggesting that differences in functional diversity (and not just in functional evenness) were at the root of the observed pattern. According to BTA, recovery of functional diversity at the dredged sites has not yet occurred (Table 2).

3.2.7. Rao's quadratic entropy coefficient

Rao's Quadratic Entropy coefficient (Q) combines elements of diversity and dissimilarity into a measure of functional diversity based on multiple traits (Petchey and Gaston, 2002; Mason et al., 2003; Ricotta, 2005). Due to its dependence on an abundance-based diversity measure – which can lead to counterintuitive behaviour of the index (Botta-Dukát, 2005; Cooper et al., 2008b) – and because of the observed surge of *Balanus crenatus* in 2004 at one of the Reference sites, the index has been calculated with species presence/absence data, thus eliminating the potentially confusing influence of wildly fluctuating abundance information. A Rao's Q value is obtained for each trait represented in a sample. These individual trait values can be averaged to give an overall Q value for the sample. Rao's Q was consistently high at Reference sites (Fig. 3i), but never significantly different from that at Low dredging intensity sites. However, at the High dredging intensity site there were significant differences from Reference conditions in all years except 2002. There were no significant differences between High and Low intensity sites. Multivariate analysis of Rao's Q values per trait by sample revealed three statistically distinct assemblages (Fig. 4f). Two of the statistically distinct assemblages were present at dredged sites, whereas the third was exclusive to the Reference sites. There was no difference in Rao's Q values between years but significant differences in Rao's Q values between Reference sites and dredged sites (ANOSIM $R > 0.29$, $p < 0.01$). No difference in Rao's Q was found between High and Low intensity sites (ANOSIM $R = 0.04$, $p = 0.06$). Although significant, the average dissimilarity between the assemblages at Reference and dredged sites was low (3.14%). The relative contribution of the top 5 traits to the difference in functional diversity observed between Reference and dredged sites is presented in Table 1. According to the SIMPER test, a greater functional diversity within the top 4 traits was observed in the assemblage at the Reference sites than at the dredged sites and these differences in functional diversity accounted for over 71% of the dissimilarity between the two assemblages. Despite the high degree of similarity in Rao's Q values between different treatments, the functional diversity of assemblages at both dredged sites has still not recovered to pre-dredged levels.

3.3. Assemblage description

Judging by all the results presented above, it would appear there are at least two distinct benthic macrofaunal assemblages represented around Dredging Zone 2 in Area 408, one inhabiting undisturbed Reference sites and the other inhabiting dredged sites. The assemblage inhabiting the Reference sites appeared to be the most stable, as all but one of the indices tested with multivariate analyses revealed that the assemblage was indistinguishable from one year to the next. BTA supported its distinction from other sites but also showed a change in trait composition over time. Multivariate analyses on sediment particle composition did not detect a temporal change at Reference sites, with sediments remaining mixed throughout the study period. The assemblage at the Reference sites was characterised by a high abundance and diversity of organisms (96 species, the most characteristic being *Urothoe elegans*, *Polycirrus* sp., *Notomastus* sp., *Urothoe marina* and *Mysella bidentata*). The families responsible for most of the production at Reference sites were the Upogebiidae, Terebellidae, Capitellidae and Urothoidae. All trophic groups were well represented, but most organisms belonged to the suspension or detrital feeding guild (TG1), indicative of an undisturbed environment. Traits displaying the greatest diversity at the Reference sites were size, feeding habit and body form.

The assemblage inhabiting dredged sites did not appear to have been affected differently by high or low intensity dredging; instead, according to most indices the assemblage composition appeared indistinguishable and relatively stable over time regardless of dredging intensity. However, sediment particle composition was very different from Reference conditions, appearing more homogeneous through the loss of its gravel and silt fraction. It also appeared to have changed over time at both High and Low dredging intensity sites, possibly influencing the diversity of function displayed by the re-colonising and constantly adapting assemblage. The assemblage was much poorer in abundance, species number and diversity than at Reference sites (46 species, the most characteristic being *Ophelia borealis*, *Polycirrus* sp., *Nephtys cirrosa* and *Bathyporeia elegans*). Most of the production at dredged sites was accomplished by the families Nephtyidae, Opheliidae, Terebellidae and Orbiniidae. Again, all guilds were represented, but the most conspicuous were the surface detritus feeders (TG2) and subsurface deposit (or specialist environment) feeders (TG4), indicative of degraded conditions. Size, relative mobility and body form were the most diverse traits observed at dredged sites but their diversity remained lower than that at Reference sites.

4. Discussion

The distinction between the two assemblages identified above seems to stem from the lasting physical effects caused by sustained dredging and screening of the seabed, given that they are separated by virtue of being found either in areas that have been dredged or in undisturbed areas. Differences between the assemblages inhabiting dredged and undisturbed areas are manifest in more ways than the simple difference in the number of organisms and species present in each area. Biomass, species diversity, species evenness, as well as functional diversity and functional evenness also seem to have been affected by dredging. Correlations between the patterns of all these variables and the pattern of sediment composition were all statistically significant. There are ecological and practical consequences of such findings, both in terms of assemblage recovery and management of aggregate extraction activities.

4.1. Recovery

Biological recovery has traditionally been assessed by a return to the same faunal assemblage present at a site prior to disturbance. Since in most instances knowledge of faunal assemblage structure before disturbance is lacking, an accepted practice is to use a similar but undisturbed neighbouring area as a benchmark against which the affected site can be compared (e.g., Boyd and Rees, 2003; Boyd et al., 2005; Cooper et al., 2007a, 2008b; Barrio Froján et al., 2008). This approach, however, has inherent limitations that must be acknowledged. Firstly, this form of recovery may not always be a realistic prospect, as the seabed is a dynamic environment which can result in a constantly fluctuating or progressively changing faunal assemblage. This, in turn, alters the population available to re-colonise a disturbed area of seabed (Matthews et al., 1996). Ellis

(2003) believes it would be very difficult to demonstrate recovery convincingly, since equilibrium cannot be reached in a continuously changing ecosystem. Secondly, if the physical nature of the seabed has been permanently altered it may no longer be able to accommodate its original assemblage (e.g., Desprez, 2000). So, under the definition given above, recovery will never occur unless the seabed structure is returned to pre-disturbance conditions. During the 4 years of the present investigation, the seabed at the designated reference sites did not appear to change significantly in either sediment particle composition (Fig. 4a) or in its macrofaunal assemblage composition (Fig. 4b). Therefore, using the assemblage at Reference sites as a benchmark against which to measure recovery at the nearby dredged sites seems perfectly legitimate.

Based on the results from univariate analyses conducted for the present investigation (summarised in Table 2), recovery of functional diversity at the Low dredging intensity site was apparent before 2001, less than 2 years after the cessation of dredging according to the TD and Rao's Q indices. According to the ITI, a return to conditions similar to those at Reference sites occurred at the Low intensity site in 2004 (5 years post cessation), yet the mean ITI score was still below the 60 score threshold indicative of a disturbed habitat. The fact that undisturbed assemblages are naturally close to the 60 score threshold diminishes the utility of the ITI as a reliable predictor of recovery or, indeed, as an accurate measure of climax functional diversity in coarse sedimentary habitats. All other indices and metrics suggested that a return to undredged conditions had not occurred at the Low intensity site by 2004, therefore taking more than 5 years to recover. A similar conclusion can be reached based on comparisons between the Reference and High intensity site, where again, only the ITI and TD appeared to have returned to values similar to those at Reference sites. All multivariate analyses revealed an assemblage at the Low intensity site that was most similar to that at the High intensity site and that recovery of functional diversity had not occurred at either site by 2004 (5 years after cessation of dredging).

What is also apparent is that despite the considerable difference in intensity between High and Low sites, dredging, however limited, has had a remarkably similar and persistent effect on the assemblage, making it indistinguishable (at least statistically and by all measures) between both dredged sites. Even differences between samples falling under the ITI categories 'modified' and 'degraded' were not significant. Therefore, it could be the case that it is the initial act of dredging and not the persistence or intensity of dredging that has the most damaging effects on the macrofaunal assemblage. In other words, once the initial damage is done, prolonging the damaging activity makes little further difference to the recovery process.

It is difficult to determine whether this is actually the case at Area 408, because the practice of sediment screening is likely to have a confounding influence on conditions at the dredge sites. The High and Low dredging intensity sites were in close proximity to each other, therefore, any fine sediment returned to the seabed through screening will have settled beyond the exact location from which it was dredged (Hitchcock and Drucker, 1996). Fine sediments dredged and screened at the High intensity site are likely to have settled over a relatively wide area, probably covering parts of the Low intensity site. It is perhaps the wider-reaching effects of sediment screening that are responsible for some of the assemblage conditions observed at the Low intensity site, effects that are disproportionate to the perceived degree of direct disturbance. Indeed, the sediment profile at the Low intensity site was more similar to that at the High intensity site than it was to the Reference site (Figs. 1 and 4a), despite having been subject to relatively little direct dredging.

Given that dredging and screening are likely to have altered the physical environment by reducing the proportion of the coarser components of the sediment, recovery of functional diversity to a level that is indistinguishable from that of a undredged state is unlikely to occur, as organisms that require the stability afforded by coarse sediment can no longer become established. Future assemblages at dredged sites may eventually be as productive and functionally diverse as those at undisturbed sites (as is already becoming apparent from some results of univariate analyses), and are likely to become as diverse as undisturbed areas of similar sediment composition, but they will never be able to acquire exactly the same taxonomic assemblage or functionality profile as that present where there is coarse sediment (clearly supported by results from multivariate analyses). After 5 years of recovery, an approximation to levels of biological and functional diversity found at reference sites has not been observed in dredged sites at Area 408.

4.2. To screen or not to screen

The implications of the findings of this study on how the practice of aggregate dredging is regulated are profound. Most aggregate extraction licences stipulate that the seabed must be left in a state similar to that in undisturbed sediments (DEFRA, 2002, 2007). Clearly, after years of dredging and screening at Area 408, this condition has not yet been met and, given the altered physical state of the seabed, it is unlikely that it ever will be met.

The issue of sediment screening appears to be a particularly important one. In a similar exercise to that undertaken in this study, Cooper et al. (2008b) investigated the rate of recovery of functional diversity at a relinquished aggregate extraction site in the English Channel (Area X in the Hastings Shingle Bank). In Area X sediment screening was not permitted, therefore all sediment dredged from the seabed was retained, effectively minimising any change in the sedimentary profile of the remaining seabed. Recovery of a low dredging intensity site according to all indices calculated took place between <5 and 7 years after the cessation of dredging. It would seem that full recovery of the studied aspects of assemblage composition and functional diversity is possible after dredging providing screening has not taken place.

Based on this knowledge, it may be necessary to re-visit the guidelines for the conduct of aggregate dredging to either alter the process of how aggregate is collected and processed, or to accommodate the fact that recovery to an identical or equivalent functional state may not always take place after sustained screening. Given the current findings, alternative measures of remediation of areas already disturbed by sustained screening must also be considered if existing licence conditions are to be met (e.g., habitat restoration or mitigation by permanent exclusion from other areas). There have already been studies on the feasibility and effectiveness of some remediation measures (e.g., Cooper et al., 2007b), and further research into more efficient uses of all dredged material is necessary in order to reduce the need for screening in the future.

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Paper #7

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Gravel seeding – A suitable technique for restoring the seabed following marine aggregate dredging?

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Restoration of offshore marine habitats is a relatively new concept, with attempts in the European Union being largely instigated by requirements of various strategic directives. In this experiment, we investigate the practicality and effectiveness of gravel seeding, using a commercial aggregate dredging vessel, in order to recreate a gravel habitat. The experimental design consisted of a Treatment and Control site, both within an area of historic dredging characterised by an overburden of sand, and a gravel dominated Reference site. All sites were surveyed, using a combination of acoustic, camera and grab techniques, 2 months before, and then at 0, 12 and 22 months after the deposition of 4444 m^3 of gravel dominated sediments within the Treatment site. Although financial and practical constraints limited replication of the Treatment to one area, and so precluded strong statistical conclusions, our results suggested that the technique was both practically feasible, and successful in terms of returning the physical and biological attributes at the Treatment site to a state more representative of gravelly substrata in the wider, un-impacted environment.

1. Introduction

Physical impacts associated with marine aggregate dredging are well documented and include changes in seabed topography (Dickson and Lee, 1972; Shelton and Rolfe, 1972; Millner et al., 1977; van der Veer et al., 1985; Kenny and Rees, 1994; Newell et al., 1998; Boyd et al., 2002), and in the composition of seabed sediments (Poiner and Kennedy, 1984; Kenny et al., 1998; Desprez, 2000; Boyd et al., 2005; Newell et al., 2004; Cooper et al., 2005; Desprez et al., 2010).

Changes in the composition of seabed sediments are most commonly associated with the practice of sediment screening (Hitchcock and Drucker, 1996; Newell et al., 1998, 2004; Sutton and Boyd, 2009), whereby unwanted sediment fractions, usually sands, are returned to the seabed. Newell et al. (2004) estimate that between 20 and 80% of the material dredged can be rejected overboard during the screening process, depending on customer requirements and the ratio of sand to gravel in the dredged deposits. In addition to screening, fine sediments are also lost from the dredger as a result of 'overspill', as water within the dredge hopper is displaced through overspill chutes as the cargo is loaded (Newell et al., 1998; Sutton and Boyd, 2009). Investigations into the fate of sediment plumes resulting from screening and overspill suggest that the vast majority of material falls out within the immediate vicinity of the dredger, with suspended sediment levels decreasing to background levels within 200–500 m (Poiner and Kennedy, 1984; Hitchcock and Drucker, 1996). However, as settled plume material could be re-suspended and moved further on subsequent tides, the potential secondary impact zone may be larger (Newell et al., 2004; Cooper et al., 2006).

The level of impact of the processes described above on benthic communities is documented to be highly dependent on the characteristics of a given site (Desprez, 2000; Boyd and Rees, 2003; Sutton and Boyd, 2009). For example, it is suggested that deposition of fine, sandy sediments may have less of an impact at sites characterised by mobile sands, with faunal communities adapted to high energy environments, than on those where stable gravels predominate and faunal communities are comprised of relatively high percentages of encrusting epifaunal species (Desprez, 2000; Boyd and Rees, 2003).

In England and Northern Ireland, marine aggregate extraction is licensed under the Marine Minerals Regulations (UK, 2007). Within this statutory framework, the Marine Mineral Guidance 1 (ODPM, 2002) states that "dredging should aim to leave the seabed in a similar physical condition to that present before dredging started. This measure is designed to enhance the possibility of, and rate at which, the seabed recovers physically and biologically to its pre-dredging condition". There is, however, presently little understanding,

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and no statutory definition of what 'similar' means in this context. This is problematic for the industry regulator, who, against a backdrop of various strategic directives which call for restoration of impacted environments (e.g. Article 2 of the EEC Habitats Directive (Council Directive 92/43/EEC), the EC Water Framework Directive (Directive 2000/60/EC), the EC Marine Strategy Framework Directive (Directive 2008/56/EC) and Article 2 of Annex V to the OSPAR Convention (OSPAR 1992)), is required to make judgements about whether observed changes to the seabed on cessation of dredging are acceptable.

With the exception of a recently published study looking at the potential for use of waste shell material from the shellfish processing industry for seabed restoration (cf. Newell and Garner, 2007), most examples in the literature involving the addition of substrata to the seafloor are concerned with capping projects, with the primary objective of isolating contaminated sediments from the overlying water column (Polayes, 1997; Bona et al., 2000; Mohan et al., 2000; Moo-Young et al., 2001; Simpson et al., 2002). Few studies have investigated patterns and rates of recolonisation of capped sediments, and those that do exist are largely concerned with effects of biota on the integrity and effectiveness of the capping layer with respect to isolation of contaminants (Bona et al., 2000).

In this experiment, we investigate the practicality and effectiveness of gravel seeding, using a commercial aggregate dredging vessel, in order to recreate a gravel habitat. To assess the effectiveness of the technique we set up two hypotheses. Firstly, that 'if gravel is seeded onto an area of seabed characterised by an overburden of sand, then the proportion of exposed gravel will increase', and secondly that, 'an increase in the proportion of exposed gravel will, given sufficient time, result in a change in the faunal community to one more typical of gravelly habitats in the wider environment'.

2. Materials and methods

2.1. Study site

Licence Area 408 is located in an area known as the Coal Pit, 100 km east of the Humber estuary in the southern North Sea (Fig. 1). Water depths range between 22 and 33 m (lowest astronomical tide), and the tidal ellipse is orientated in a NW-SE direction. Maximum spring tidal velocity reaches 1.0 ms^{-1}, and the residual tidal direction and subsequent sediment transport is predominately to the south-east.

As a condition of the extraction license, the site was subdivided into a number of discrete zones, thus limiting the geographical scale of environmental impact during any one period and minimising disruption to fishing or other activities. Dredging in one of these zones (zone 2) ceased in 2000, following the removal of 1,459,131 tonnes over a period of four years between 1996 and 1999. Zone 2 occupies an area of 2.56 km^2.

Newell et al. (2002) estimate that screening at Area 408 resulted in an annual rejection of approximately 285,000 tonnes of material, some 53% of that dredged (figures based on data from 1997 to 2000). In addition, Newell et al. (2002) and Evans (2002) have shown evidence for the persistence of this material on the seabed in an area extending for at least 2 km along the axes of net sediment transport towards the south-east. The presence of this material appears to be responsible for a suppression of biomass (Cooper et al., 2005; Robinson et al., 2005) and species richness and abundance (Cooper et al., 2005) in the area.

2.2. Experimental design

The experimental design comprised a Treatment (T), Control (C) and Reference (R) site (Fig. 1). Each site consisted of a rectangular 'box' measuring 100 m × 250 m (see Table 1 for co-ordinates). Treatment and Control boxes were positioned, using historic information on the location and intensity of dredging, such that both sites had been subjected to the same intensity of dredging. In addition, previous surveys indicated both sites were characterised by surficial sands which have resulted from the screening of dredged cargoes (Newell et al., 2002; Evans, 2002). The Treatment box was orientated in a NW-SE direction, in line with the expected tidal axis, to allow the dredger undertaking the seeding operation to hold position against the prevailing tide. The Reference site, located to west of zone 2 and remote from the effects of dredging,

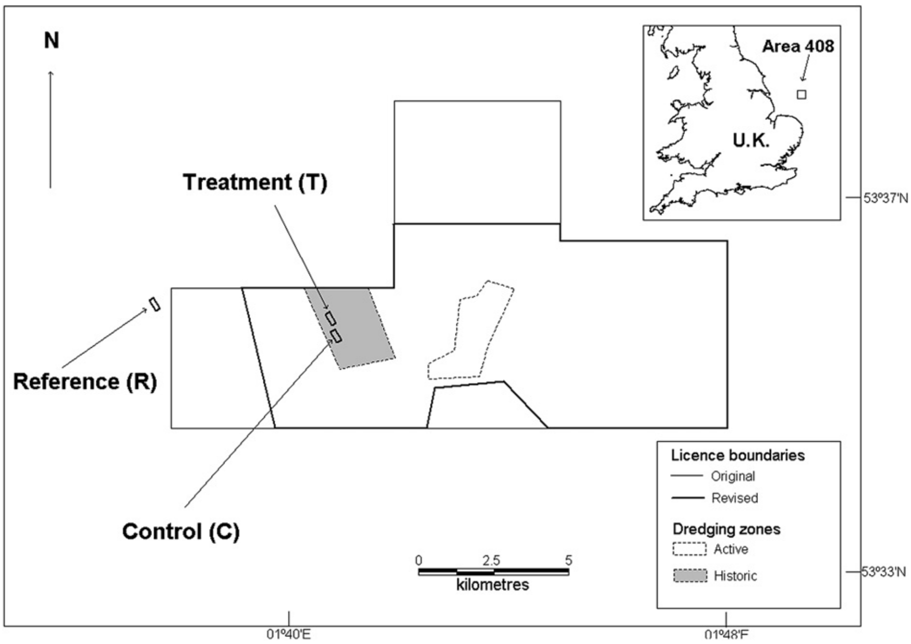

Fig. 1. Map showing the location of Area 408 and the position of Treatment (T), Control (C), and Reference (R) boxes. Also shown is the active dredging zone from where the seeded gravel was dredged.

Table 1
Co-ordinates of sites and numbers of 0.1 m^2 Hamon grab samples collected during each of the surveys (t−2, t0, t+12 and t+22).

Site	Node	Box co-ordinates		Number of samples collected			
		Latitude	Longitude	t−2	t0	t+12	t+22
Treatment (T)	A	53° 35.733′N	01° 40.654′E	10	10	10	10
	B	53° 35.761′N	01° 40.732′E				
	C	53° 35.644′N	01° 40.846′E				
	D	53° 35.617′N	01° 40.767′E				
Control (C)	A	53° 35.550′N	01° 40.748′E	10	10	10	10
	B	53° 35.557′N	01° 40.826′E				
	C	53° 35.461′N	01° 40.939′E				
	D	53° 35.433′N	01° 40.861′E				
Reference (R)	A	53° 35.887′N	01° 37.489′E	10	10	10	10
	B	53° 35.915′N	01° 37.568′E				
	C	53° 35.798′N	01° 37.681′E				
	D	53° 35.771′N	01° 37.603′E				

was characterised by surface gravel, and was chosen to be representative of likely pre-dredge conditions. For consistency, this site was also orientated in a NW-SE direction.

All three sites were surveyed on four separate occasions. The initial baseline survey was undertaken in May 2005, prior to the deposition of gravel within the Treatment box. Sites were then surveyed on three further occasions: immediately post deposition in July 2005, one year after deposition in July 2006, and nearly two years after deposition in May 2007. Each of the surveys was labelled according to the period of time, in months, before or after the deposition (i.e. t−2, t0, t+12 and t+22).

During each survey, a 0.1 m^2 Hamon grab was used to collect samples from 10 randomly positioned stations within each site, for later analysis of macrofauna and sediment particle size (see Table 1). Station positions were re-randomized during each survey. In addition, the status of the Treatment and Control sites was assessed using multibeam bathymetry, and all three sites were assessed using underwater video. The purpose of these surveys was to assess whether the seeding of gravel within the Treatment box had increased its similarity, in terms of sediment particle size and faunal composition, to that of the Reference site. The Control site allowed some sort of assessment of whether the changes observed within the Treatment site were a result of the gravel seeding, or natural variability in the near vicinity of the treatment site.

Limited space at the study site meant that it was not possible to replicate either the Treatment or Control site. We recognise that this limits our survey design in that we have only one true replicate of the treatment application and one of the two control applications. Thus, our conclusions, in a strict statistical sense, are limited only to assessing differences that occurred between the treatment and control areas – and may not necessarily reflect what would happen over the wider dredged area (see Hurlbert, 1984). However, whilst we recognise that it is difficult to draw strong statistical conclusions from our design, there is much that we can discover. In particular, we can assess whether the hypothesised effects occur at the sites we did sample; we can also assess the extent to which the seeding works in terms of practicality.

2.3. Gravel seeding

The aim of the gravel seeding operation was not to replace the entire volume of material dredged, merely to return a thin layer of coarse material to the surface of the seabed in order to promote faunal recovery. To achieve 100% coverage of the Treatment box, and to reduce the potential for smothering by the surrounding sand, it was decided to aim for a capping layer of approximately 15–20 cm depth. This required a total of 5000 m^3 or 9000 tonnes of deposit (based on a density of 1.8 tonnes m^{-3}). A typical dredger operating at this licence holds around 4000 tonnes per cargo and hence it would require at least two full loads to achieve the required depth. With two full cargoes, a back calculation suggests a theoretical deposit depth of 17.7 cm. Given the vessel's cargo discharge rate of 16 tonnes min^{-1}, it was estimated that discharge of 4444 m^3 of material would take 8.33 h. An assumption was made that the method of deposition from the vessel, using the stern conveyor, would allow a 'footprint' of 2 m on either side of the deposition track. For this reason, deposition tracks were planned to be 4 m apart.

2.4. Survey methods

2.4.1. Acoustic surveys

A Kongsberg EM3000 dual head multibeam echosounder was used to collect detailed bathymetry data from the area. Data was acquired using Kongsberg SIS software, processed using Caris HIPS, and visualised in IVS3D Fledermaus. The latter software was then used to calculate changes in the volume of material on the seabed.

2.4.2. Video surveys

Photographic surveys, using underwater video and stills techniques, were conducted using a Simrad™ video camera and a Benthos DSC™ digital stills camera. Both devices were mounted on a towed camera sledge, which was towed at approximately 1.85 km h^{-1} through each box. These surveys were used to obtain additional ground-truth information on the physical and biological status of the seabed.

2.4.3. Hamon grab surveys

Sediment samples were collected using a 0.1 m^2 Hamon grab. Following retrieval of the grab, the contents were released into a 60 L plastic bin and a 500 ml sub-sample was removed for later analysis of particle size composition (see Boyd, 2002). The remaining sediment, following initial processing to remove the <1 mm sediment fraction, was preserved in a formosaline solution and returned to the laboratory for identification and enumeration of fauna (see Boyd, 2002).

2.5. Data analysis

Samples were assigned a factor combining SITE and TIME (T t−2, T t0, T t+12, T t+22, C t−2, C t0, C t+12, C t+22, R t−2, R t0, R t+12, R t+22). Analyses were then carried out to explore differences between the data we had from the Treatment, Control and Reference sites in order to give an indication of whether the results we obtained were those that we would have expected. Due to the limited replication of the study we have generally refrained from using formal hypothesis tests because of their limited applicability for our data.

2.5.1. Sediment particle size

A data matrix consisting of untransformed particle size data (percentage gravel, coarse sand, medium sand, fine sand and silt/clay) was analysed using the PRIMER 6® package (Clarke and Warwick, 1994). Data were averaged according to the factor SITE/TIME, prior to production of a correlation based Principle Components Analysis (Chatfield and Collins, 1980). The technique places samples in a 2-dimensional space corresponding to their relative similarities, based on the measured variables. The axes of the plot are referred to as Principle Component 1 (PC1) and Principle Component 2 (PC2).

2.5.2. Macrofauna

The following univariate indices were calculated for each faunal sample: number of species (S) (including colonial taxa), number of individuals (N) (excluding colonial taxa), biomass (AFDW g) and Pielou's eveness (J′). Values of Pielou's eveness describe how evenly individuals are distributed among the different species. Values range from 0 to 1, with higher values reflecting a more even distribution (i.e. less dominance).

For each of the univariate measure, a sample variogram (see Cressie, 1991) was plotted for various points in time to see whether there was a correlation between the distance apart of the samples and the value of the measure. Where there is no spatial correlation then the use of the sample replicates to compare the treatment and control sites is more robust. This analysis was carried out using the software R (R Development Core Team, 2008).

Multivariate community analyses were carried out using the PRIMER 6® package (Clarke and Warwick, 1994). Species abundance data (colonial taxa excluded) were averaged using the factor SITE/TIME before being fourth-root transformed to down-weight the importance of very abundant species. Similarities were then calculated between each SITE/TIME combination, using the Bray–Curtis coefficient of similarity (Bray and Curtis, 1957), prior to production of a multidimensional scaling ordination (MDS) plot. Using individual sample data, the similarity percentages routine (SIMPER) was used to identify the level of 'within-group' sample similarity, and also the species responsible for the similarity/differences within/between sites during each sampling occasion.

3. Results

3.1. Deposition process

The gravel seeding operation took place from 19th to 21st July 2005 using the aggregate dredging vessel *M.V. Arco Axe*, a 98 m, 3498 tonne, trailer suction hopper dredger. In total, two 2222 m^3 cargoes were dredged from within an active zone of Area 408 and deposited with the Treatment box. Both cargoes were screened heavily to maximise the gravel content.

The dredging process for each cargo took between 8 and 9 h and was followed by approximately 4.5 h of draining or 'de-watering' prior to discharge. The majority of both cargoes were discharged using the same process as that employed during normal offloading operations in port. This involved moving material up a ramp and into a large hopper at the stern of the vessel using two large buckets, which are pulled across the surface of the cargo by large steel cables (Fig. 2a). Once in the hopper, material was fed out over the stern of the vessel via two conveyer belts (see inset Fig. 2a). When discharging to a dredge wharf, the last conveyer would be rotated through 90°. As the vessel offloaded its cargo, it effectively lost ballast, and, as a result of the poor weather conditions encountered at the time, a decision was made to switch to a 'wet' discharge for the remainder of both cargoes (approximately 800 tonnes). In this process, which is usually reserved for 'emergency disposal' of a cargo, the dredge pipe is submerged slightly and water pumped into the hold. At the same time, six hydraulically operated doors are opened in the bottom of the hull and any material flows out (Fig. 2b).

High winds (Force 6–8) prevented the vessel from running a series of pre-planned deposition lines. We therefore used an alternative approach whereby the vessel discharged whilst moving through the Treatment box under influence of wind, tide and vessel power. By displaying the vessel track on the ships plotter, offset to the discharge point, the vessel was manoeuvred to produce as even a coverage over as possible (see Fig. 3).

3.2. Acoustic surveys

Data from multibeam bathymetric surveys were used to assess topographical changes on the seabed following deposition. The modelled seabed surface from the baseline survey (t−2) was subtracted from those of post-depositional surveys (t0, t+12 and t+22) to assess changes in seabed bathymetry over time (Fig. 4). Adverse weather conditions during the baseline survey (t−2) account for the parallel lines (ENE-WSW) running across the boxes. A clear increase in seabed height within the Treatment box was visible immediately following deposition (t0), whereas no significant change was

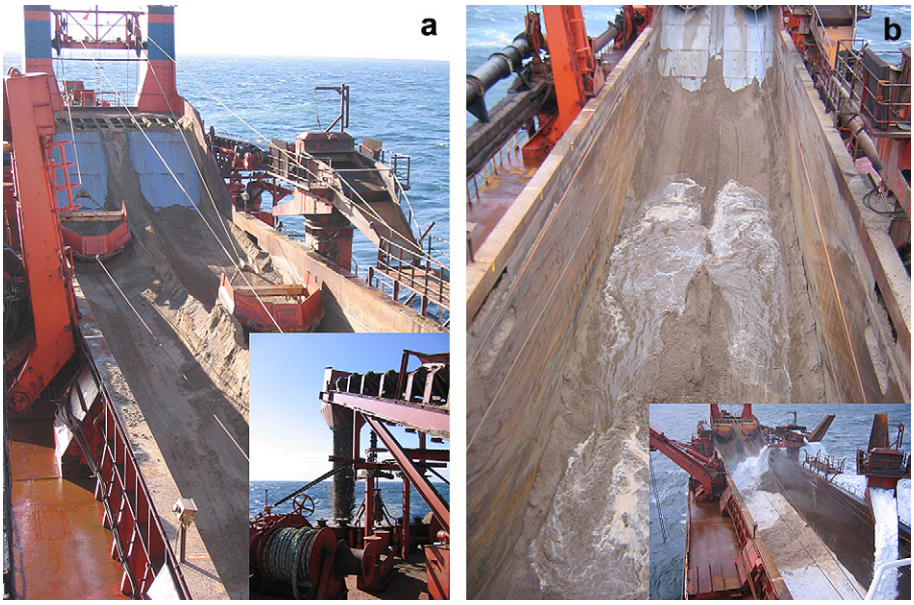

Fig. 2. a) Buckets moving material into hopper at the stern of the vessel during 'dry' discharge. Inset shows material being deposited over the stern of the vessel. b) Flooding of the cargo hold following opening of hold doors during 'wet' discharge of remaining cargo.

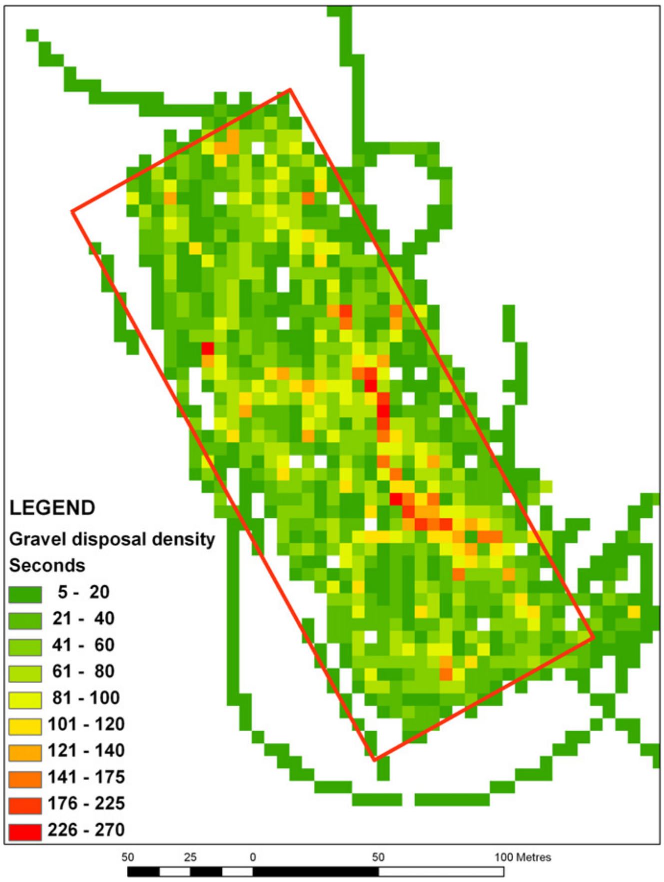

Fig. 3. Plot of vessel track (offset to discharge point) during discharge of material. Colour scale indicates density of coverage.

observed in the Control box. Within the Treatment box, multibeam bathymetry data suggests that a volume of almost 4897 m^3 was added following disposal of the gravel to the seabed. This is in line with the estimates of the amount of gravel that was disposed to the seabed by the dredger. The observed reduction in volume of the deposited material on the seabed during the latter phase of the study (4641 m^3 at t+12, 3438 m^3 at t+22) may be largely explained by a gradual compaction of the deposited material over time. Within the Control box, small changes in sediment volume may be explained by natural variability and/or errors associated with the adverse weather conditions during the baseline survey.

3.3. Underwater video

Fig. 5 shows three images from each of the Reference, Control and Treatment sites at t−2, and a further three images from the Treatment site at t0, t+12 and t+22. Throughout the period of study, the Reference site was characterised by gravel and shell, with a well-developed epifaunal community including hydroids, sea anemones and bryozoans. In contrast, the Control site was characterised by mobile sandy sediments, possibly the result of past screening activity (Evans, 2002), with very little exposed gravel or obvious epifauna. Similarly, conditions did not change at this site over the period of investigation.

During the baseline survey, the Treatment site was indistinguishable from the Control site. However, at t0, immediately post deposition, whilst some small patches of thin sand cover were observed, there was an obvious increase in the quantity of exposed gravel within the Treatment site. One year later (t+12), conditions remained similar, although sand ripples were evident within small sand patches, particularly in the south of the site. In the final survey (t+22), areas of exposed gravel were again evident within the Treatment site, together with some obvious epifauna (e.g. hydroids and the common starfish *Asteria rubens*). There was some evidence

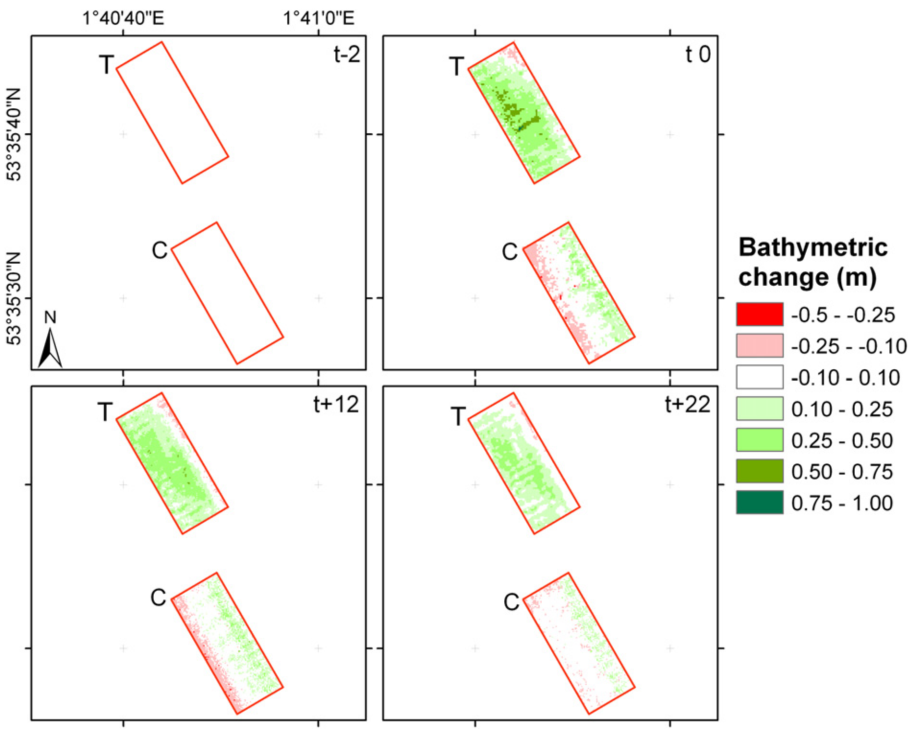

Fig. 4. Change in seabed bathymetry within Treatment and Control boxes. Plots are based on a comparison of baseline bathymetric data (t−2) with subsequent surveys (t0, t+12 and t+22).

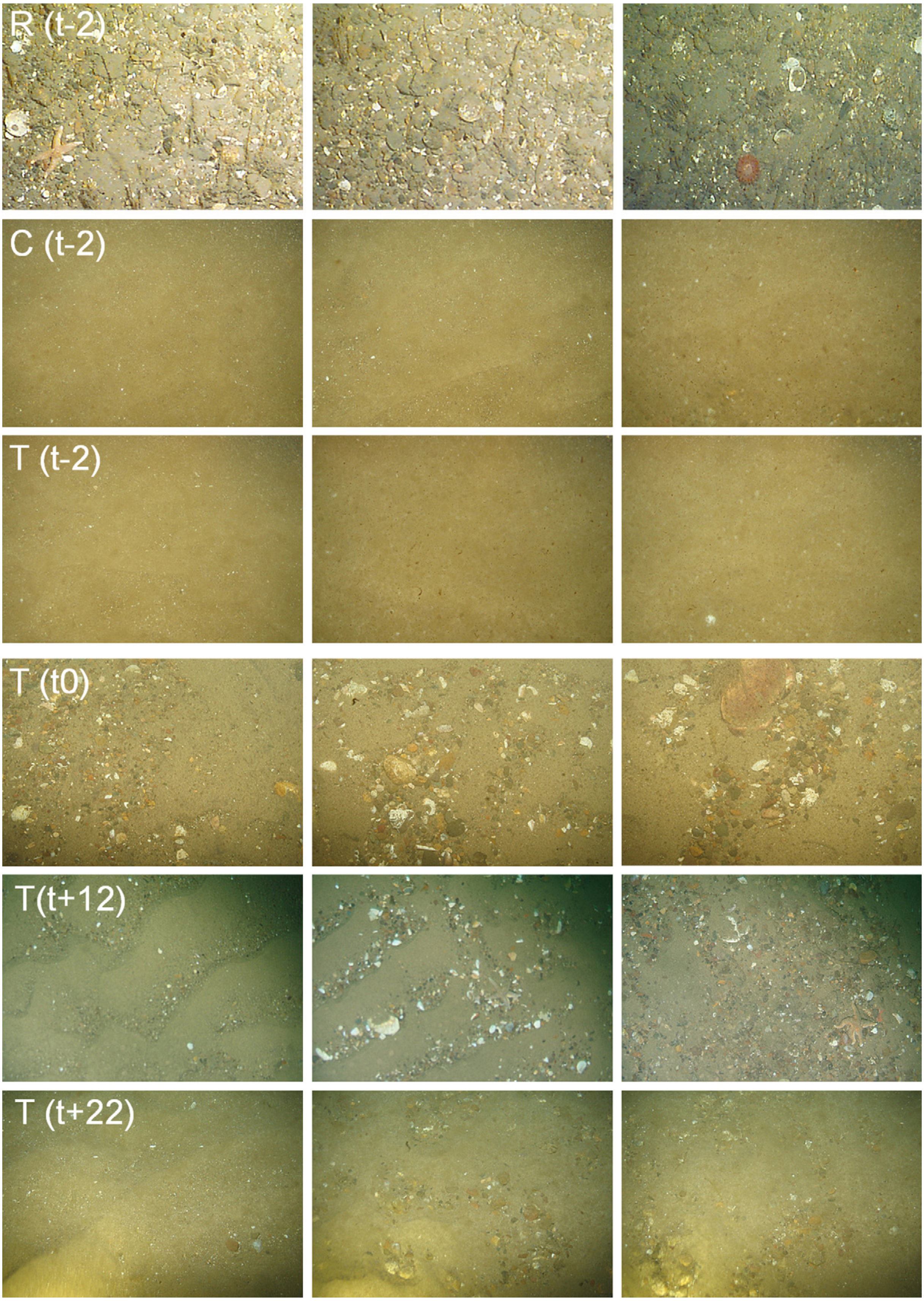

Fig. 5. Images of the seabed acquired from camera tows from through the Treatment, Control and Reference sites. Part a) shows all three sites during the baseline, pre-deposition survey (t0). Part b) shows only the Treatment site during the 3 post-depositional surveys (t0, t+12 and t+22). Images from the Reference and Control site post-depositional surveys are not shown as no obvious changes were apparent from the baseline survey (t0).

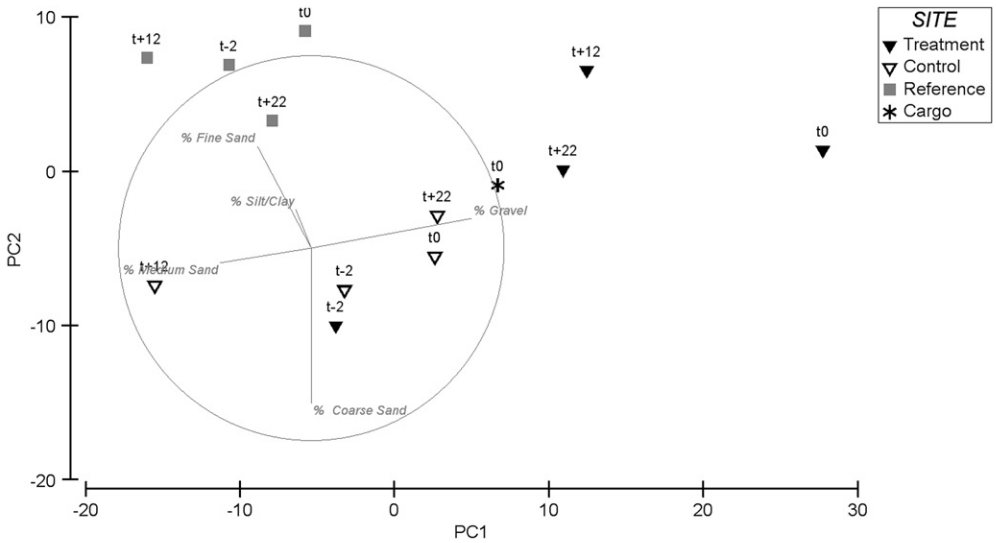

Fig. 6. PCA ordination of samples, based on untransformed sediment particle size data and averaged by the factor SITE/TIME.

of a slight increase in the quantity of surficial sand present within the Treatment site at this time, although this assessment was made difficult by poor visibility.

3.4. Sediment particle size

Fig. 6 shows a PCA ordination of samples, based on untransformed sediment particle size data and averaged by the factor SITE/TIME. PC 1 is characterised by an increase in gravel and accounts for 77.6% of the variability. PC 2 is characterised by a decrease in coarse sand and an increase in fine sand and accounts for 20.9% of the variability. In addition, summary data are shown in Table 2.

Despite the clearly higher gravel content of surface seabed sediments at the Reference site, as shown by the underwater video survey during the baseline survey (t−2), sediments from all sites were, in broad terms, similar in sediment particle size composition (Table 2). This suggests that there are differences in sediment stratification between sites, with coarser material underlying the surficial sands present within the Treatment and Control sites. This explains the position of Reference samples in the top left corner of the PCA ordination.

The particle size composition of cargo sediments was based on an analysis of two ‘representative’ sub-samples obtained from the 2nd cargo. At 48.4%, the gravel content of the cargo samples was higher than samples taken from both the Treatment (38.4%) and the Control sites (39.4%) (Table 2).

At t0, immediately following deposition, the Treatment sample centroid had moved upwards and to the right within the PCA plot, indicating an increase in the proportion of gravel (67%) and a corresponding decrease in the proportion of coarse and medium sands. During subsequent surveys (t+12 and t+22), the centroid position for the Treatment samples remained in the top right of the PCA plot, indicating that samples continued to be characterised by a higher percentage of gravel.

3.5. Macrofauna

3.5.1. Univariate

For each of the univariate measures, we examined the spatial correlation between samples within the same site and at the same time. During t+22, the semi-variograms for the Treatment site (not shown) gave some indication that the closest together samples were the least different, although we found very little evidence of this for the other sampling occasions. Our general conclusions were that the samples were fairly spatially independent. This means that we can have more confidence that similarities in the points from

Table 2
Mean values (±SD) for sediment particle size characteristics at each site (T, C and R) during each survey (t−2, t0, t+12 and t+22). Also shown are the results for the cargo.

Site	Time (months)	Mean (mm)	Sorting	Skewness	Kurtosis	% Gravel	% Coarse Sand	% Medium Sand	% Fine Sand	% Silt/Clay
T	t−2	1.6 (±0.6)	2.2 (±0.3)	−0.4 (±0.5)	2.0 (±1.0)	38.4 (±12.2)	28.5 (±11.7)	23.9 (±6.9)	9.1 (±2.0)	0.1 (±0.2)
	t0	4.0 (±2.7)	2.0 (±0.1)	0.7 (±0.6)	3.0 (±1.9)	67.0 (±12.5)	18.2 (±6.5)	10.0 (±4.6)	4.5 (±2.2)	0.3 (±0.2)
	t+12	2.5 (±1.3)	2.2 (±0.2)	0.2 (±0.6)	2.1 (±0.6)	55.4 (±17.3)	14.4 (±3.4)	17.1 (±8.7)	12.9 (±7.3)	0.3 (±0.3)
	t+22	2.1 (±0.6)	2.3 (±0.2)	0.1 (±0.4)	1.9 (±0.4)	52.8 (±10.3)	19.8 (±5.7)	17.5 (±6.0)	9.8 (±3.2)	0.2 (±0.2)
C	t−2	1.8 (±1.0)	2.2 (±0.4)	−0.4 (±0.6)	2.1 (±1.1)	39.4 (±16.2)	26.5 (±11.2)	23.8 (±9.6)	10.2 (±3.6)	0.1 (±0.2)
	t0	2.0 (±0.7)	2.4 (±0.3)	−0.2 (±0.5)	1.8 (±0.5)	44.9 (±12.0)	24.2 (±10.5)	21.9 (±8.0)	8.8 (±3.0)	0.1 (±0.2)
	t+12	1.3 (±1.0)	2.0 (±0.4)	−0.8 (±0.8)	3.1 (±1.8)	30.5 (±18.2)	24.6 (±5.3)	33.1 (±11.8)	11.7 (±3.5)	0.1 (±0.2)
	t+22	2.0 (±0.7)	2.4 (±0.1)	−0.2 (±0.4)	1.6 (±0.2)	45.8 (±10.9)	21.7 (±5.1)	22.5 (±7.6)	9.9 (±2.3)	0.1 (±0.2)
R	t−2	0.9 (±0.5)	2.5 (±0.3)	0.1 (±0.2)	2.9 (±0.7)	35.3 (±10.2)	14.7 (±1.4)	26.0 (±4.8)	19.6 (±4.7)	4.4 (±2.6)
	t0	1.1 (±0.4)	2.5 (±0.3)	0.1 (±0.2)	2.6 (±0.3)	40.5 (±7.9)	12.4 (±1.5)	25.2 (±3.7)	19.3 (±2.9)	2.7 (±0.6)
	t+12	0.8 (±0.3)	2.5 (±0.3)	0.1 (±0.2)	3.2 (±0.5)	31.1 (±7.7)	14.1 (±1.9)	28.9 (±3.7)	21.0 (±3.0)	5.0 (±1.3)
	t+22	1.2 (±0.9)	2.3 (±0.2)	0.0 (±0.5)	2.9 (±0.5)	37.8 (±14.6)	16.7 (±3.6)	27.0 (±7.0)	15.9 (±8.0)	2.6 (±2.2)
CARGO	t0	1.7 (±0.5)	2.3 (±0.1)	0.1 (±0.3)	2.3 (±0.1)	48.4 (±7.8)	21.6 (±3.9)	17.6 (±3.3)	11.6 ±1.0)	0.8 (±0.1)

the same site in the multivariate plots are not due to spatial dependence – and do represent some sort of replicate variability over the Treatment/Control site.

A total of 20,284 individuals, belonging to 279 taxa were identified from the 120 × 0.1 m^2 Hamon grab samples analysed during this study. In addition, 53 colonial species were also identified.

Clear differences in the distribution of species, individuals and biomass between sites were apparent (see Table 3). For example, over the duration of the study the Reference site was host to a total of 298 species (90% of the total number identified), 13,617 individuals (67% of the total abundance) and 26.95 g of biomass (64% of the total). Crustaceans were the numerically dominant component of the macrofauna at this site, accounting for 57% of the assemblage. They were followed by polychaetes (25%), 'others' (9%), molluscs (8%) and echinoderms (1%).

In contrast, only 120 species (37% of the total), 1348 individuals (7% of total abundance) and 1.78 g of biomass (4% of the total) were found at the Control site. Polychaetes (51%) were the dominant component of the macrofauna at this site, followed by molluscs (18%), echinoderms (12%), crustaceans (11%) and 'others' (8%).

At the Treatment site, numbers of species and individuals were initially similar to the Control site. However, one year after deposition (t+12), numbers of taxa increased by a factor of 1.66, abundance by a factor of 9.56, and biomass by a factor of 37.55 relative to the Control site. In addition, the assemblage changed from one dominated by polychaetes, to one dominated by crustaceans. Despite a decline in the numbers of species and individuals seen at the Treatment site during t+22, the number of species was still 28% higher, numbers of individuals 280% higher, and biomass 198% higher than the Control site.

Fig. 7, plots a–c show the mean values for the number of species, individuals, and biomass (AFDW) found at each site over the period of investigation. In contrast to the richer gravel reference site, lower mean values of all univariate measures were consistently found at the Control site. Mean values at the Treatment site were initially similar to the Control site. However, one year after deposition (t+12), mean values for the number of individuals and biomass increased to values similar to the Reference site. Reasons for the decline in all univariate measures observed at the Treatment site during the final survey (t+22) are unclear, although a similar trend was observed at the Reference site. Despite this decline, mean values of all univariate measures remained higher than at the Control site at t+22.

Fig. 7d shows mean values of Pielou's eveness (*J*'), based on abundance data, at each site over the period of investigation. Consistently high mean values of *J*' with low variance, reflecting a consistent even distribution, were seen at the Control site. In contrast, mean values of *J*' at the Reference were lower, and, on occasion, more variable. The Treatment site was initially similar to the Control site (high mean *J*', low variance), then, following deposition, showed a marked increase in variability, and a decrease in mean values of *J*' (reflecting an increase in dominance, more typical of the Reference site).

3.5.2. Multivariate

Fig. 8 shows an MDS ordination of samples averaged by the factor SITE/TIME. The close proximity of the Treatment and Control sample centroids from the baseline survey (t−2) indicates a high degree of similarity in terms of faunal composition. Samples were characterised by a sparse assemblage, typical of mobile sandy sediments (e.g. the Echinoderm, *Echinocyamus pusillus*; Molluscs, *Polinices pulchellus and Thracia villosiuscula*; Polychaetes, *Ophelia borealis and Polycirrus*; and the Crustacean, B*athyporeia elegans).*

Whilst many of these species were also present at the Reference site, this location was characterised by a much greater range of species, including many epifaunal species (e.g. Hydroids, *Hydrallmania falcata, Halecium,* and *Sertularia*; Bryozoans, *Alcyonidium diaphanum* and *Electra pilosa;* Polychaetes, *Pomatoceros lamarcki* and *Clymenura*; and the Crustacean, *Balanus crenatus).* Many of these species, which require a hard surface for attachment, were either absent or much less frequently encountered at the previously dredged sites.

Results from t0, immediately post deposition, revealed a similar picture in terms of the differences between the sites sampled at t−2. Of particular interest was the continued similarity between the Treatment and Control sites.Given that the seeded gravel was largely devoid of fauna, this suggests that a number of species, also found within the Control site, were able to quickly colonise the deposited gravel.

Twelve months after deposition (t+12), the MDS plot shows that the Treatment site had shifted away from the Control site and towards the Reference site. In addition, the results of a SIMPER analysis showed that whilst levels of similarity between the Control and Reference have remained constant between t0 and t+12 (17%), similarity between Treatment and Reference site has increased from 14% to 28%. Changes within the Treatment site over time result from an increase in the occurrence of species more commonly associated with the Reference site (e.g. Crustaceans, *Balanus crenatus* and *Urothoe elegans*; Polychaetes, *Scalibregma inflatum, Lanice concheliga, Scoloplos armiger* and *Pectinaria koreni;* the Hydroid, *Sertularia sp.*; and the Mollusc, *Ensis* sp.), and a decrease in species more typical of the Control site (e.g. the Mollusc, *Goodallia triangularis*; and the Polychaete, *Nephtys cirrosa).*

During t+22 the difference between Treatment and Control sites was less obvious. Nevertheless, the increases in similarity between the Treatment and Reference site from t+12, were maintained, and, despite the recolonisation process being incomplete, the data (whilst realising its limitations) support the hypothesis that an

Table 3
Total number of species (S), individuals (N) and biomass (g AFDW) found at the Treatment (T), Control (C)and Reference (R) sites during each survey (t−2, t0, t+12 and t+22). The number of colonial taxa contributing to the total number of species is shown in parentheses. Also shown are the total number of species, individuals and biomass found at each site over the duration of the study.

Metric	Site	t−2	t 0	t+12	t−22	Total
No. of species (S) (Total: 328)	T	46 (7)	63 (11)	118 (18)	97 (11)	161 (23)
	C	55 (9)	64 (11)	71 (8)	76 (8)	120 (17)
	R	162 (25)	190 (33)	203 (37)	190 (33)	298 (49)
No. of individuals (N) (Total: 20284)	T	222	473	3081	1543	5319
	C	249	372	322	405	1348
	R	2427	2024	5231	3933	13617
Biomass (g AFDW)	T	0.60	2.45	7.51	2.81	13.37
	C	0.42	0.22	0.20	0.94	1.78
	R	7.51	3.54	9.60	6.30	26.95

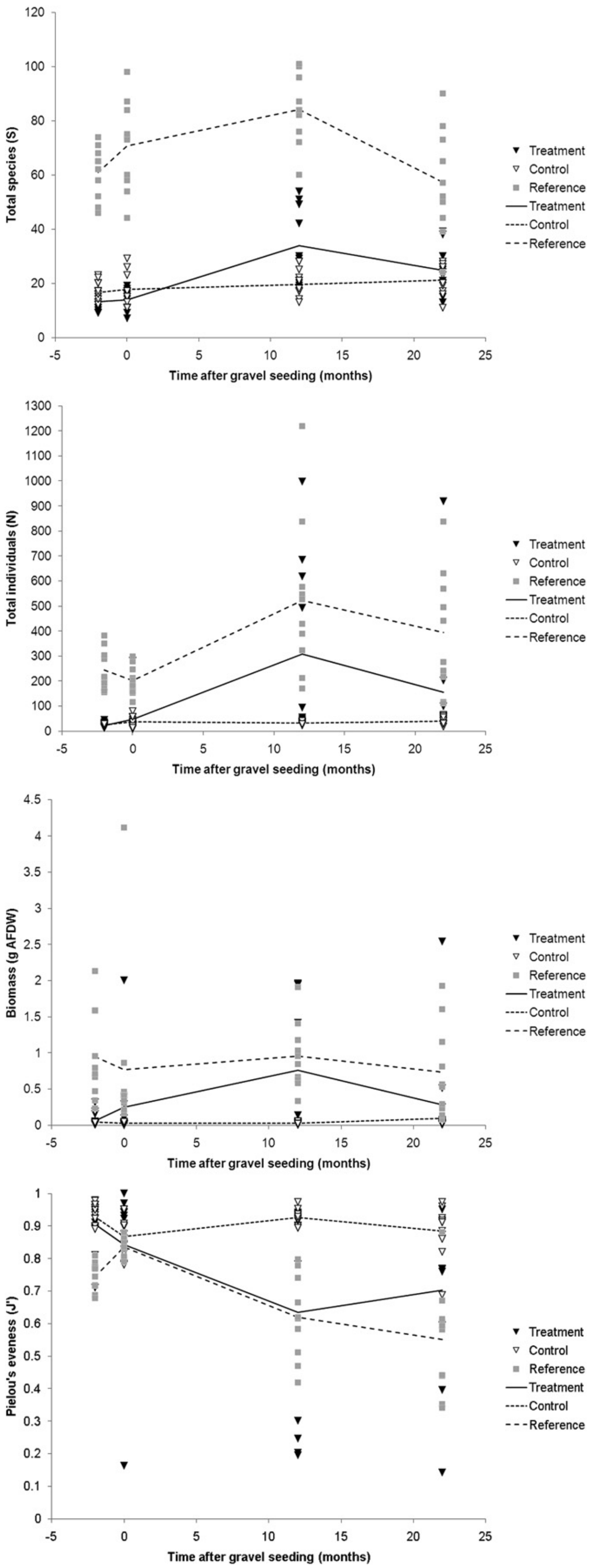

Fig. 7. Scatter plots for a) number of species (S), b) number of individuals (N), c) biomass (g AFDW) and d) Pielou's eveness (J') (based on abundance data), found at the Treatment (T), Control (C) and Reference (R) sites over the period of investigation. Mean values at each site are shown by lines drawn between sampling events.

increase in the proportion of exposed gravel will, given sufficient time, result in a change in the faunal community to one more typical of gravelly habitats in the wider environment.

4. Discussion

4.1. Practicality

Results from this study have shown that it is possible to use a commercial aggregate dredging vessel, typical of those operating on UK licensed extraction areas, to undertake gravel seeding. Although not specifically designed for such operations, material can be deposited using the conveyor belt at the stern of the vessel, the same technique employed to offload sand and gravel in port. Additionally, a through-the-hull method, usually reserved for emergency disposal of the cargo, can also be employed. A combination of these techniques allows disposal to take place in a range of weather conditions.

In this study we adopted a strategy of moving within the Treatment box under influence of wind, tide and vessel power. This reduced the amount of time necessary for the vessel to be outside the box in order to make turns, and, irrespective of weather conditions, may be the most pragmatic approach where the size of the treatment area is limited. Whilst the approach could result in a less even coverage, the risk can be reduced by periodically stopping the discharge when the vessel passes over areas of seabed already seeded.

The use of commercial dredging vessels has the advantage that licence holders themselves have the capacity to undertake this form of remediation, without the need to bring in additional equipment or contractors. In addition, assuming suitable deposit material can be sourced locally, vessels could undertake restoration whilst on-site during the course of their normal activities, reducing the cost of such operations.

4.2. Effectiveness

Whilst the lack of replication in the survey design means that we are unable to draw firm conclusions about a generalised effect of gravel seeding, this study shows strong evidence for the existence of a treatment effect. Further support for this judgement is provided by Cooper et al. (2005), who observed that, in the four yeas running up to the current study (2001–2004), sediments within Area 408, zone 2 were characterised by mobile sands with a sparse faunal assemblage.

4.2.1. Physical

The hypothesis that seeding gravel onto an area of seabed characterised by an overburden of sand would increase the proportion of gravel exposed at the surface of the seabed was supported by the data, with a clear difference between the Treatment and Control site after deposition. Despite this, there was still more sand evident at the surface of the seabed within the Treatment site, compared to the Reference site.

The most likely explanation for the presence of sand, in addition to the newly deposited gravel within the Treatment site, was the significant quantities of this sediment fraction within the cargo. Indeed, data showed that the proportion of gravel within the cargo was only 10% higher than in samples taken from the Treatment and Control sites prior to deposition (t−2). Despite this, seeding resulted in a 22% increase in the mean gravel content of seabed

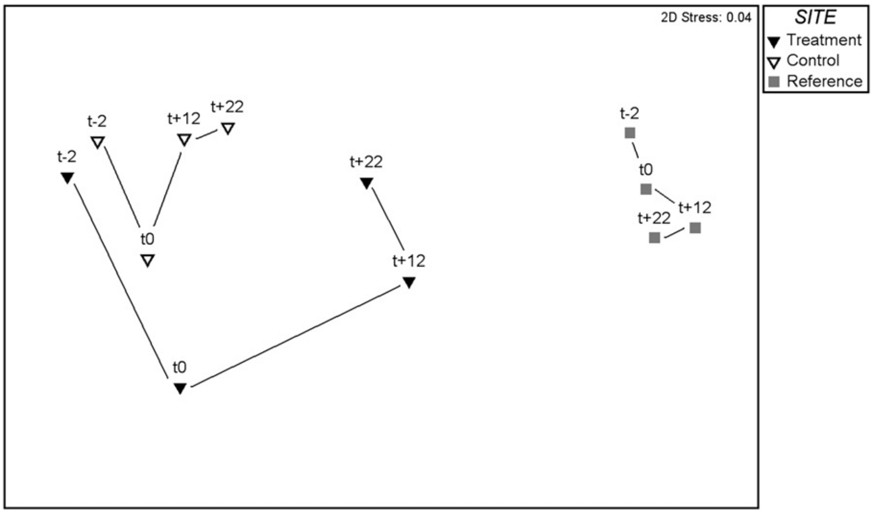

Fig. 8. MDS ordination of Bray–Curtis similarities from 4th root transformed species abundance data (colonial species excluded), averaged by the factor SITE/TIME.

sediments, relative to the Control site. This suggests that useful results can be achieved with only a marginal increase in gravel content of the seeding material. Reasons for the better than expected result may have been that the gravel dropped straight down, whilst sand had a wider dispersion (Poiner and Kennedy, 1984; Hitchcock and Drucker, 1996). Whilst this could be seen as compounding the problem elsewhere, the quantities involved are likely to be relatively minor relative to any previous impact.

Although the physical effect was still clearly present at t+22, the data indicate that there may have been a lessening of the effect over time, with an increase in sand cover within the Treatment site. We suggest this result may have occurred due to mobilisation of sand from the surrounding seabed, as a result of the adverse weather conditions in the days before and during the final survey. In addition, the effect may have been amplified by the decline in height of the deposit on the bed, making it more vulnerable to coverage with sand. This illustrates the need for continued monitoring at this site to assess the long-term success of the operation. It also highlights the need to determine the feasibility of such projects in relation to site characteristics (e.g. physical environment, hydrodynamic conditions, geological conditions and sediment characteristics (Mohan et al., 2000)).

4.2.2. Biological

Previous studies on the recolonisation of disturbed gravel sediments suggest that the process typically takes between 2 and 12 years, and in some certain cases even decades (Collie et al., 2009; Foden et al., 2009). The range of recovery times is explained by differences in the composition and age structure of the pre-impact or reference community (Newell et al., 2004), and differences in the time required for physical recovery of the seabed. Given that the current study spans a period of only 22 months after gravel seeding, it is probable that the process of recolonisation is ongoing. This is made more likely given the isolation of the Treatment site within an area of sparsely populated sandy deposits. As a result, colonisation by species not previously present within the site is likely to result largely from planktonic settlement, as opposed to migration of adults/juveniles from surrounding deposits (cf. Newell et al., 1998).

Despite the above, early indications were positive, with the changes evident following the seeding operation being consistent with the expected patterns of colonization and succession (cf. Newell et al., 1998). For example, results showed an increase in the numbers of species, individuls and biomass, one year after deposition (t+12). These changes resulted from an increase in the occurrence of species more commonly associated with Reference site, and reflect an increase in habitat complexity brought about by the gravel seeding. One of the key early colonisers, responsible for much of the increase in abundance and community dominance seen at t+12 was the barnacle, *Balanus creanatus*. This species has been shown to be an opportunistic coloniser of gravels following the cessation of marine aggregate dredging (Boyd and Rees, 2003; Cooper et al., 2007).

Without further monitoring, it is impossible to know whether the declines observed at the Treatment site resulted from adverse weather conditions at the time of the final survey; similar declines at the Reference site support this hypothesis. Further support for this idea comes from Collie et al. (2009) who hypothesised that burial, by migrating sand, of the gravel within their experimental recolonisation trays may have inhibited survival of colonial epifauna. An alternative explanation for the declines is that they are typical of a community entering an equilibrium phase (cf. Newell et al., 1998). Nevertheless there remained a biological enhancement in the community at the end of the study, although the high variability, probably reflecting the patchy physical effect, has the effect of masking this. This illustrates the problem of trying to exactly recreate a habitat (Hawkins et al., 2002), and highlights a need to set realistic targets for restoration (Grayson et al., 1999). In view of this, Elliott et al. (2007) suggest this form of remediation would be more appropriately termed 'enhancement' rather than restoration.

The fact that the biological enhancement is associated with the seeded gravel suggests that unless the physical conditions can first be restored, either naturally, or, as in this study through a process of intervention, impacted sites may not fully recover. As such, simply removing the stressor, an approach to recovery advocated by a number of authors (e.g. Hawkins et al., 2002; Elliott et al., 2007) may not always achieve the desired result.

5. Conclusions

More explicit licence conditions, in terms of what constitutes an acceptable seabed condition at the licence term, will help focus monitoring efforts during the life of the licence. This would help industry and the regulator to intervene, at an early stage, thus reducing the likelihood that unacceptable conditions will result at the end of the licence term.

Although financial and practical constraints limited replication of the Treatment to one area, and so precluded strong statistical conclusions, this experiment has shown that it is technically possible to partially restore the seabed in response to the formation of a persistent overburden resulting from screening. We suggest a suitable next step might be to initiate a larger study, capable of drawing conclusions about a generalised effect of gravel seeding. It might also consider different approaches to restoring the composition of seabed sediments (e.g. different depths of gravel deposit, use of the surface layer of sediment from a new dredge zone, preferentially removing sand through a process of reverse screening, or by dredging the impacted area to source gravel which can then be returned to the surface of the seabed).

Whether it makes sense to try to restore the composition of sediments where marine aggregate dredging has resulted in the formation of an overburden of sand will depend on the results of a detailed site-specific feasibility assessment, which would need to establish the following: (1) thickness and extent of the overburden resulting from dredging, (2) potential for natural recovery, (3) significance of the changes for the health of the wider ecosystem and other legitimate interests, (4) quantity of material required for restoration, versus the total quantity of material extracted from the site prior to cessation of dredging (clearly the two figures ought to be very different), (5) source and nature of material to be used for gravel seeding, and any requirement for screening, (6) impact of screened sediments on restoration efforts, (7) the likelihood of long-term success, taking into account local conditions, (8) the financial and environmental costs and benefits of restoration.

Whilst Area 408 provided a useful site to establish our experiment, these questions are clearly beyond the scope of the current study. Therefore we make no assessment of the merit of undertaking restoration at this site. Future case studies will help determine if this form of restoration is, in general terms, a viable proposition.

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Assessment of ecosystem function following marine aggregate dredging

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ABSTRACT

A number of indices designed to assess ecosystem function were applied to an existing benthic macrofaunal dataset collected following recent marine aggregate extraction activity at the Hastings Shingle Bank (UK). The objective of the study was to assess the use of these different functional metrics, some incorporating biological traits analysis, to investigate the rate of recovery in ecosystem function after dredging impact. All of the indices tested behaved in a broadly similar fashion following the aggregate extraction event, although some suggested faster rates of functional recovery than others. All indicated that the disturbed area of seabed was capable of full recovery given enough time. It is considered that this outcome may be because the physical nature of the seabed was unlikely to have been permanently altered by dredging for aggregate by the method used. This is not always the case following aggregate extraction and depends on the dredging protocol used (e.g., sediment screening). The indices tested (some applied for the first time to benthic macrofaunal data) were considered to be complementary to traditional environmental assessment metrics and each might be used under different circumstances.

1. Introduction

Exploitation of the marine habitat takes many forms and the sustainable exploitation of the marine environment is of increasing global concern (Gray, 2005). There is arguably more difficulty in assessing aquatic than terrestrial systems since direct observation of the former is difficult (Paterson, 2005; Solan et al., 2006). Sand and gravel is routinely dredged from the seabed around the globe to supplement land-based aggregate sources for the construction industry, or as a source of material for beach nourishment and coastal defence (Singleton, 2001). In the UK, the British Government regulates the marine aggregate industry in order to mitigate the effects of dredging on the environment by imposing a stringent set of conditions. These conditions are usually site-specific and can cover issues such as the boundaries of extraction areas and extraction rates, how the seabed must be left at the cessation of dredging, and the precise environmental attributes that must be monitored before, during and after dredging (DEFRA, 2002, 2007). The laudable desire by Government to help minimise long-term environmental impact has resulted in a number of initiatives to characterise and assess the rate of recovery of the seabed after disturbance (Leung et al., 2005; Gray et al., 2006; Kwok et al., 2008). The present investigation represents one such initiative and reports on comparisons between traditional and alternative measures of ecosystem recovery.

Traditionally, biological recovery has been assessed by a return to the same faunal assemblage present at a site prior to disturbance or by comparison of the affected site with a suitable reference site (e.g., Boyd et al., 2003, 2005; Cooper et al., 2007). Metrics used to assess recovery typically include biodiversity analysis such as the numbers of species and/or individuals in an assemblage. However, this approach presents a number of challenges. Firstly, this form of recovery may not always be a realistic prospect, especially when the physical nature of the seabed has been altered and it can no longer accommodate its original assemblage (e.g., Desprez, 2000). In addition, the seabed can be a dynamic environment, resulting in a constantly fluctuating or progressively changing faunal assemblage, which in turn affects the populations available to re-colonise a disturbed area of seabed (Matthews et al., 1996). Furthermore, Ellis (2003) upholds that it would be very difficult to demonstrate recovery convincingly, since equilibrium cannot be reached in a continuously changing ecosystem. For these reasons, and because it is important to know how ecosystems work, it may also be sensible to consider the functional capacity (or health) of the ecosystem rather than simply the range and proportion of species present.

Benthic organisms perform a number of ecosystem-level processes, often described as 'ecosystem functions'. These functions encompass any process of transformation, whether measurable or not, that occurs in an ecosystem. They include all metabolism, catabolism

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and dynamic processes such as sediment bioturbation or active re-suspension, as well as the production and transfer of food, oxygen, and nutrients, the recycling of waste material and the sequestration of harmful substances. Whilst some ecosystem functions can be undertaken by a variety of different organisms, it is generally believed that a greater diversity of species increases the stability and resilience of an ecosystem's capacity to perform its various functions (Cardinale et al., 2000, 2002). Linked to this belief is the notion of functional redundancy in ecosystems, where the loss of a species belonging to one functional group may not affect the basic functioning of the ecosystem, since the function performed by that species is taken up by another member in the same functional group (Fonseca and Ganade, 2001). The extent to which species can be lost before basic ecosystem processes are compromised depends on the functional richness (i.e., the number of functional groups) and evenness (i.e., the distribution of species across functional groups) in an ecosystem (Mouillot et al., 2005). In terms of the impact on ecosystem function following dredging for aggregate, an area of dredged seabed with an altered physical character may, in time, accommodate an altogether different assemblage to that of its original, pre-dredged state, but have recovered all functional capacity. This concept allows for an ecosystem to be altered without immediately reaching the conclusion that it has been permanently damaged.

A number of approaches have been developed to characterise the richness and evenness of ecosystem function (Díaz et al., 2004). These techniques are seen as complementary to the traditional indices that simply capture species diversity, yet to date, very few studies have used them, especially in the context of the marine environment. To this effect, the purpose of the present investigation was twofold: firstly, to identify a number of functional indices/approaches suitable for use with an existing marine faunal assemblage dataset, and secondly, to compare the results of these techniques against traditional measures of assemblage composition (i.e., numbers of species and of individuals). All this is set in the context of assemblage recovery after dredging for aggregate, with a view to improving our understanding of the effects of this activity on the wider ecosystem.

2. Methods

One area of recent aggregate extraction that has undergone extensive environmental monitoring is the Hastings Shingle Bank, situated 10 km south of Hastings in the eastern English Channel (Cooper et al., 2007). The relatively homogeneous nature of the gravel deposits at this site, together with the condition imposed by Government that no sediment screening could take place, make it an ideal model to monitor the biological recovery of the seabed after disturbance by dredging, without the added complication of the sediment composition being altered. The dredging activity removes the surface layers of sediment, causes local sediment redistribution and disturbs surface structure and habitat. Cooper et al. (2007) have already reported on the rate and extent of biological recovery based on traditional metrics after different dredging regimes. A summary of their findings is included in Table 4. The present investigation uses the same dataset analysed by Cooper et al. (2007) to test and compare the behaviour of selected functional diversity indices.

2.1. Sampling Design

Two areas of seabed previously subjected to relatively high (H) and low (L) levels of dredging intensity were identified on the Hastings Shingle Bank (Fig. 1). Two reference (i.e., undredged - R) areas were also selected for comparative purposes. All four sites were monitored annually over the period 2001–2004, using a combination of acoustic, video and grab sampling techniques (see Cooper et al., 2007 for

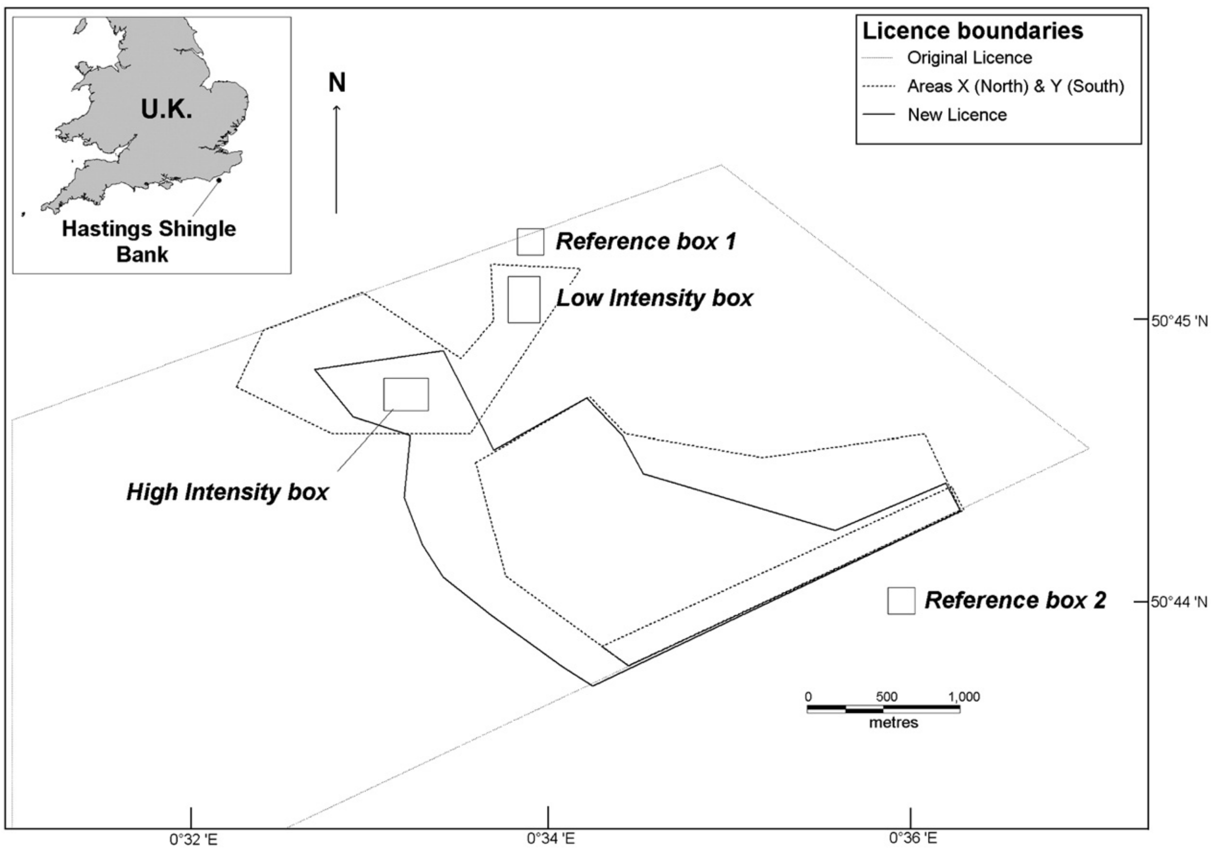

Fig. 1. Treatment boxes in relation to the boundaries of historic and new licensed aggregate extraction areas on the Hastings Shingle Bank (from Cooper et al., 2007).

detailed sampling and sample processing methodology). Since the site as a whole was last dredged in 1996, the sampling regime at inception was intended to provide a sequence of data from 5 to 8 years after cessation of dredging. However, an unexpected resumption of dredging within the High intensity site, during 2002 and 2003, allowed an additional assessment of the immediate effects and aftermath of renewed dredging on the seabed at that site. The early stages of biological recovery could then be assessed after dredging ceased.

2.2. Functional Analyses

A literature search identified 12 potential analyses that quantify ecosystem function (see Table 1). Of those 12, only five were considered suitable for use with the data available; these were the Infaunal Trophic Index (ITI), Somatic Production (P_S), Biological Traits Analysis (BTA), Taxonomic Distinctness (TD) and Rao's Quadratic Entropy coefficient (Rao's Q). Each of the selected techniques is considered in turn.

2.2.1. Infaunal Trophic Index and Trophic Group Analysis

The Infaunal Trophic Index (ITI) was developed as a tool for marine environmental monitoring in response to organic enrichment (Word, 1979; Maurer et al., 1999). It focuses specifically on organisms' feeding habits, which are thought to be one of the central mechanisms underlying ecosystem function (Pearson and Rosenberg, 1978, 1987). Trophic group analysis assigns taxa into guilds based on their shared feeding habits. The four recognised feeding guilds (or trophic groups) are defined as: (1) suspension or detrital feeders, (2) surface (or interface) detritus feeders, (3) surface deposit feeders, and (4) sub-surface deposit feeders (or specialised environment feeders). The total number of individuals in each of the four groups is entered into a formula generating a numerical value that indicates the trophic composition of the infaunal communities. Values of the index vary from 0 to 100, with values between 100 and 60 generally indicating unaffected (normal) seabed conditions (i.e., dominated by organisms of group 1), values between 60 and 30 indicating modified/ intermediate communities, and values between 30 and 0 reflecting degraded or polluted conditions (i.e., dominated by organisms of group 4). Since the ITI is dependent on Pearson and Rosenberg's (1978) model for organic enrichment, it has yet to be validated for use with respect to physical disturbance. Several researchers have used the ITI, including Charvet et al. (1998), Mancinelli et al. (1998), Desrosiers et al. (2000) and Chícharo et al. (2002), with varying degrees of success. For the present investigation, a purpose-designed Excel workbook was used to calculate ITI values for every sample using all taxa present (Codling and Ashley, 1992). Where feeding type information for individual taxa was absent, representative information from higher taxonomic levels was employed, when this was consistent across the taxonomic group. In cases where there was more than one feeding type for that taxonomic group, those taxa were not listed/used in the calculation. In addition, the data required for the calculation of individual ITI scores (i.e., trophic group-by-sample matrix), was used to undertake a Trophic Group Analysis (TGA) using multivariate analysis techniques. Data were fourth-root transformed prior to calculation of Bray-Curtis similarity.

2.2.2. Somatic Production

Somatic Production (P_S) is the quantity of matter/energy which is potentially available as food for the next trophic level (i.e., for natural predators) (Brey, 2001; Cusson and Bourget, 2005). In order to describe the turnover of a population, benthic ecologists have traditionally used the production-to-biomass ratio (P_S/B-ratio), which relates production to the average biomass present during the period of investigation. Estimates of production are derived in a stepwise approach from abundance and biomass data. Firstly, standardised biomass records are converted to energy values using published conversion factors available for each taxonomic family (Brey, 2001). Energy values are then converted to production values using Brey's Multi-Parameter P/B-Model (Brey, 1999, 2001). The final output is a value of production for each taxon, each of which can then be combined to provide a value for the total production of a sample. The conversion factor databank and Multi-Parameter P/B-Model were downloaded from the Internet (http://www.thomas-brey.de/science/virtualhandbook/).

2.2.3. Biological Traits Analysis

Biological Traits Analysis (BTA) uses a series of life history, morphological and behavioural characteristics of species present in an assemblage to indicate aspects of their ecological functioning (Bremner et al., 2006). Changes in the patterns of trait expression within assemblages - in terms of changes in the relative abundance/ biomass of taxa exhibiting the traits - can be used to indicate the effects of human impacts on ecosystem functioning (Bremner et al., 2006). The BTA approach has a number of advantages over traditional functional diversity measures, including strong links between traits and ecosystem processes (Díaz and Cabido, 2001; Coleman and Williams, 2002). BTA provides more information on the ecological functions performed by organisms in marine benthic communities than standard diversity and trophic group approaches and has the potential to be a valuable tool for investigating the effects of anthropogenic disturbance at the ecosystem-functioning level (Doledec et al., 1999; Charvet et al., 2000).

For the present investigation, trait data were obtained from a variety of published sources (e.g., the Biological Traits Information Catalogue developed by the Marine Life Information Network), as well as a fully referenced traits database made available by one of the authors (AF). Occasionally, expert judgement and/or data from the nearest phylogenetic neighbour had to be used where reliable published information was missing. Eight biological traits were

Table 1
Selected indices/approaches used to assess marine environments

Index	Name	Application Area	Objective	Reference
AZTI	The AZTI Marine Biotic Index	Europe	Response to disturbance	Borja et al. (2000)
BHQ	Benthic Habitat Quality	International	Response to disturbance	Nilsson and Rosenberg (1997)
BRI	Benthic Response Index	California Shelf	Objective index	Smith et al. (2001)
BTA	Biological Trait Analysis	Europe	Ecosystem functioning index	Bremner et al. (2003, 2006)
ITI	Infaunal Trophic Index	California	Response to organic enrichment	Maurer et al. (1999)
IQI	Infaunal Quality Index	Europe	Describe biological status	Borja et al. (2007)
MMI	Macrofauna Monitoring Index	New Zealand	Response to dredge material	Roberts et al. (1998)
Ps	Somatic Production	International		Brey, 2001; Cusson and Bourget, 2005
Rao's Q	Quadratic Entropy Coefficient	International	Ecosystem functioning index	Ricotta (2005) Botta-Dukát (2005)
Sensitivity	Sensitivity assessment	North Sea	Management plans	Hiddink et al. (2007)
SES	Sustainable Ecological Succession	Canada	Biodiversity recovery	Ellis (2003)
TD	Taxonomic Distinctness	Europe	Taxonomic index	Warwick and Clarke (1995)

selected, each trait divided into several categories (Table 2). Each species was scored for the extent to which it displayed the trait category using a 'fuzzy coding' procedure (Chevenet et al., 1994), which allowed species to exhibit trait categories to different degrees. Traits were scored from 0 to 3, with 0 being no affinity and 3 being high affinity to a trait category. Lastly, individual trait category scores were scaled so that within a trait the sum of the values over categories equalled one. This new species-by-traits matrix was multiplied by the existing species abundance-by-sample matrix to give a species trait-by-sample matrix (see Charvet et al., 1998). The species trait-by-sample matrix resulting from Biological Traits Analysis was subject to multivariate analyses (see section 2.3).

2.2.4. Taxonomic Distinctness

Taxonomic Distinctness (TD) is the average taxonomic path length between any two randomly chosen species, traced through a standard Linnean or phylogenetic classification of the full set of species captured (Clarke and Warwick, 1998, 1999). It is calculated by summing the path lengths through a taxonomic tree connecting every pair of species in the list and dividing by the number of paths. Since it attempts to capture phylogenetic diversity rather than just species richness, it is considered more closely related to functional diversity, based on the assumption that a phylogenetically diverse assemblage accommodates a more diverse range of functional traits. One of its main advantages over other indices is that it is robust to variation in sampling effort and there exists a statistical framework for assessing its departure from expectation (i.e., funnel plots - see Fig. 5). It appears to decline monotonically in response to environmental degradation whilst being relatively insensitive to major habitat differences (Warwick and Clarke, 1995). A species-by-sample data matrix with corresponding Linnean classification information for all species (in the form of an aggregation file) is required for calculating Taxonomic Distinctness. Only presence/absence information is used in its calculation. The PRIMER package (Clarke and Warwick, 2001) is used to calculate this index, as well as to test statistically its departure from expectation.

Table 2
Selected biological traits and categories used to describe the functioning of macrobenthic taxa

Traits	Category	Category Code
SIZE	<0.5 cm	1
	0.5–0.9 cm	2
	1.0–2.9 cm	3
	3.0–5.9 cm	4
	6.0–9.9 cm	5
	≥10 cm	6
LARVAL TYPE	Planktotroph	1
	Lecitotroph	2
	Direct development	3
RELATIVE ADULT MOBILITY	None	1
	Low	2
	Medium	3
	High	4
BODYFORM	Short cylindrical	1
	Flattened dorsally	2
	Flattened laterally	3
	Ball shaped	4
	Long thin, tread like	5
	Irregular	6
DEGREE OF ATTACHMENT	None	1
	Temporary	2
	Permanent	3
ADULT LIFE HABIT	Sessile	1
	Tube permanent attachment	2
	Tube semi-permanent attachment	3
	Burrower	4
	Surface crawler	5
FEEDING HABIT	Suspension/filter	1
	Scraper/grazer	2
	Surface deposit feeder	3
	Subsurface deposit feeder	4
	Dissolved matter/symbiots	5
	Detritus feeder/sandlicker	6
	Scavenger	7
	Carnivore/omnivore	8
	Parasite/commensal	9
HABITAT	Sand	1
	Mud	2
	Stone	3
	Gravel	4
	Rocks	5

2.2.5. Rao's Quadratic Entropy Coefficient

Rao's Quadratic Entropy coefficient (Q) combines elements of diversity and dissimilarity, and has been gaining credibility as a useful functional diversity index (Petchey and Gaston, 2002; Mason et al., 2003; Ricotta, 2005). It is a generalised form of the Simpson diversity index and is a measure of functional diversity based on multiple traits. It utilises information on species abundance and more than one trait, therefore, it seems to be an improvement on measures of functional diversity that rely solely on presence/absence information. An unexpected property of Rao's Q is that its value may decrease if species richness increases (Botta-Dukát, 2005). The reason for this is that functional diversity is influenced both by species-abundance based diversity and by trait differences among species. Introduction of a new species into an assemblage increases the species-abundance based diversity, while it may decrease the average dissimilarity among species. Low average dissimilarity in an assemblage is a corollary to it having a high average similarity, and a highly similar assemblage is perceived to have a smaller functional range (or diversity) than a dissimilar one. Naturally, the converse is also possible, where a decrease in species richness results in an increase in Rao's Q.

Rao's Q is calculated in two steps once the trait matrix is established. The first step results in a measure of dissimilarity between samples based on species traits, the second combines that dissimilarity with a measure of species relative abundance. The main methodological decisions to be made are how to measure the species dissimilarity and how to characterise the proportion of a species in the assemblage. For the present investigation, species dissimilarity has been calculated based on the trait overlap between different species (Lepš et al., 2006). Trait information for each species was the same as that gathered for Biological Traits Analysis (section 2.2.3). Combining the between-species dissimilarity value with the relative contribution of each species to the assemblage (based on abundance values) completes the calculation. All calculations were performed using a freely-available, purpose-built Excel macro created by Lepš et al. (2006).

2.3. Statistical Analyses

Cooper et al. (2007) generated a dataset comprising 353 species (excluding colonials) and 120 samples (10 replicate samples per site (High, Low and Reference) per year over four years). Most of the techniques outlined above produced a single index value for each sample. These values were subjected to standard univariate and multivariate statistical analyses. One-way Analyses of Variance (ANOVA) were performed using Minitab 15® to compare mean index values between sites each year. Significant differences between index means were recognised when the resulting ANOVA value was lower than 0.05. The PRIMER 6 package (Clarke and Warwick, 2001) was used for multivariate analyses, which included non-parametric Multidimensional Scaling Ordination (MDS) based on Bray-Curtis similarity, one-way Analyses of Similarity (ANOSIM) of index values between sites, the Similarity Percentages (SIMPER) routine and calculation of an index of multivariate dispersion (MVDISP). An inverse MDS ordination of Euclidian distance between normalised index-per-sample values was performed to ascertain the degree of congruence between the information provided by each functional diversity index. In order to plot the information provided by the BTA on the inverse MDS ordination plot, the species traits-by-sample matrix was summarised into a single value-per-sample using Hill's (1973) diversity index (i.e., expH' or N1).

3. Results

The results obtained from each functional analysis technique are presented below in turn, before comparing them with the results of Cooper et al. (2007) that used traditional analysis techniques.

3.1. Infaunal Trophic Index and Trophic Group Analysis

Mean ITI scores for High, Low and Reference sites fell between 65 and 85 during all four sampling years (Fig. 2a), which according to the ITI guidelines, are indicative of a normal, unaffected community (i.e., above 60). Only occasionally did ITI scores for individual samples fall below the 60-score threshold (indicative of a modified community) and these samples were recorded most frequently from the High intensity dredging site.

Infaunal abundance values for each ITI feeding category within each sample were used to compare the functional similarity in assemblages between sites (i.e. Trophic Group Analysis). Using multivariate analysis (MDS) it was determined that despite an increase of sample dispersion with increasing dredging intensity (multivariate dispersion index values: R = 0.686, L = 0.932, H = 1.382), samples from all sites overlap with one another (Fig. 3a). This would suggest that assemblages sampled from dredged sites could be as similar, in terms of functioning, as those at the Reference sites. However, ANOSIM tests (Table 3) revealed that assemblages at the High dredging intensity and Reference sites were significantly different in all years and particularly more so during the period of renewed dredging activity in 2002 and 2003 at the High intensity site (as evidenced by ANOSIM R-values closer to 1). The assemblage at the Low dredging intensity site was not significantly different from that at the Reference site at any time. This suggests that, according to the ITI, functional recovery at the Low dredging intensity site happened sometime before 2001.

3.2. Somatic Production

In 2001 the mean value of total production at the High dredging intensity site was significantly lower than the Reference sites, implying that the High dredging intensity site had yet to recover five years after the cessation of initial dredging. In contrast, the Low dredging intensity site was not significantly different from the Reference sites, indicating that the Low site had recovered in terms of total production (Fig. 2b). During renewed dredging at the High dredging intensity site in 2002 and 2003, production values decreased further, yet by 2004, mean values of total production were not significantly different between High dredging intensity and Reference

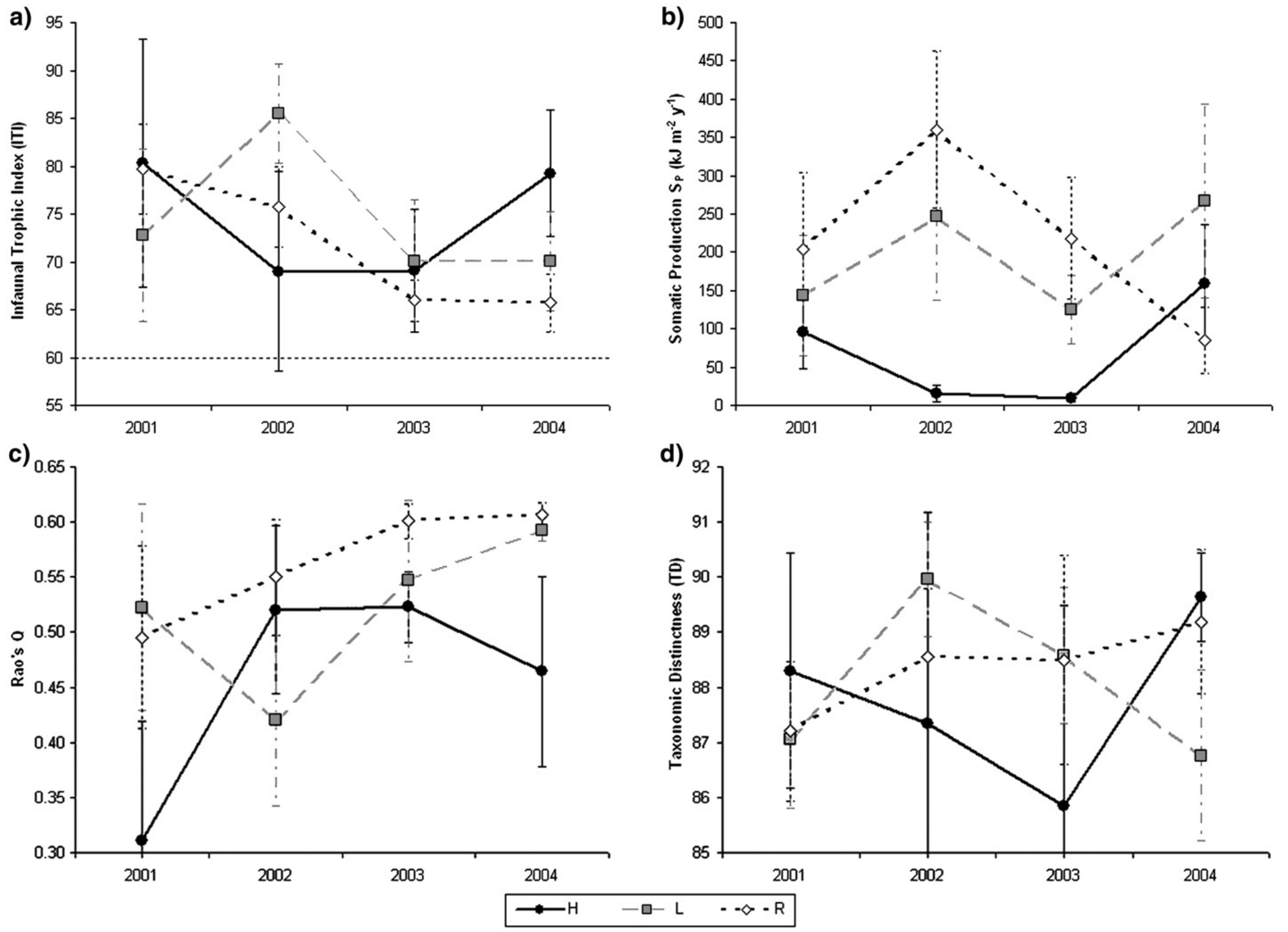

Fig. 2. Yearly mean values (±95% confidence intervals) of a) Infaunal Trophic Index (ITI) scores (horizontal dotted line at ITI value 60 indicates lower threshold of normal, unimpacted assemblages), b) Somatic Production (in kJ m^{-2} y^{-1}), c) Taxonomic Distinctness (TD) and d) Rao's Q for High and Low dredging intensity sites and Reference sites.

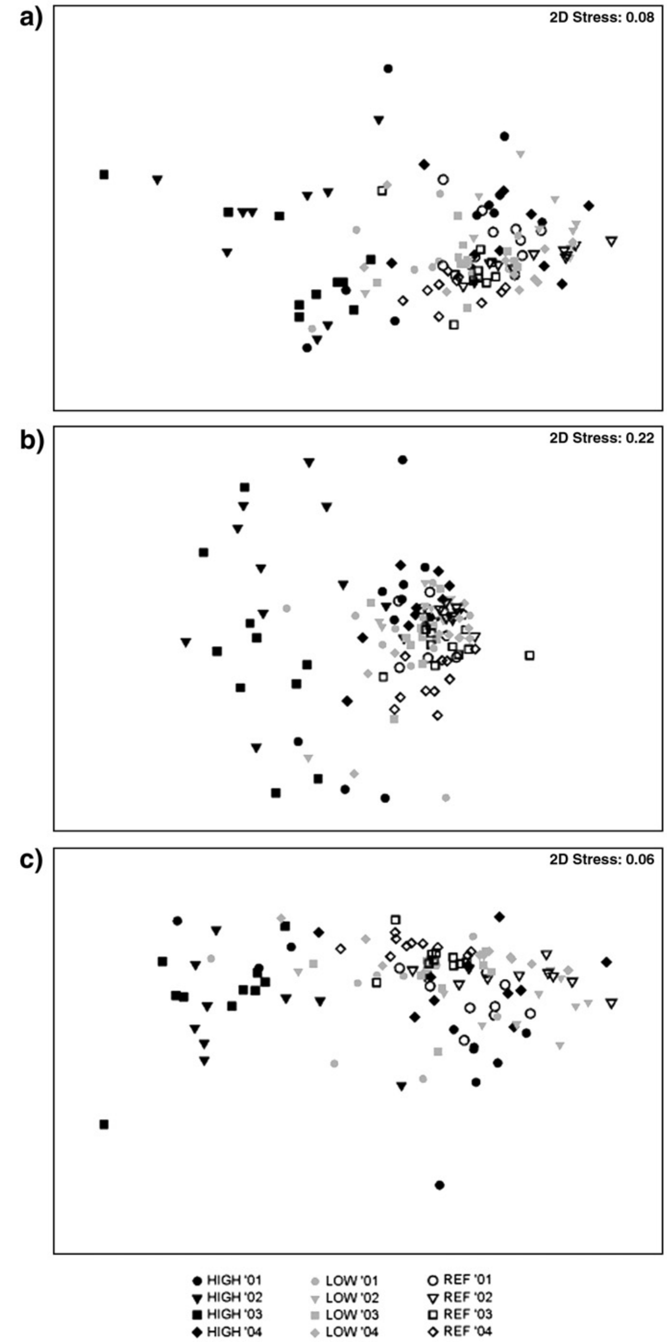

Fig. 3. MDS plot of Bray-Curtis similarities based on a) fourth-root transformed abundance values within each Infaunal Trophic Index (ITI) feeding category b) square-root transformed Somatic Production values by family and c) fourth-root transformed data obtained from Biological Traits Analysis at the High and Low dredging intensity sites and at the Reference sites from 2001 to 2004.

sites. This observation, together with the significant differences observed in 2004 between mean production values at the Low dredging intensity and Reference sites, appears to be the result of a marked decline in production at the Reference sites.

Most of the total annual production came from annelids and crustaceans, with smaller proportions contributed by lower crustaceans (barnacles), molluscs and other phyla (Fig. 4). Whilst dredging caused an obvious decline in total production, the effect was seen across all major taxonomic groups. Following the cessation of dredging, production appeared to return across all the major phyla, although initially annelids re-colonised at a faster rate than other phyla (as evidenced by their relatively large contribution to total production at the High intensity site in 2004, one year after cessation of renewed dredging).

Table 3
Summary of R-values derived from ANOSIM tests based on values calculated from different analysis techniques for samples at the High and Low dredging intensity sites and at the Reference sites

Technique	High/Ref	Low/Ref	High/Low
TGA			
2001	0.151**	0.108	0.065
2002	0.802**	0.085	0.724**
2003	0.791**	−0.008	0.718**
2004	0.339**	0.333	0.044
P_S			
2001	0.195**	0.106*	0.126*
2002	0.690**	0.111*	0.534**
2003	0.604**	0.044	0.586**
2004	0.559**	0.231*	0.255**
BTA			
2001	0.184**	0.154*	0.111
2002	0.816**	0.056	0.743**
2003	0.902**	0.044	0.854**
2004	0.388**	0.318**	0.053

* denotes significant difference at $p<0.05$, ** denotes significant difference at $p<0.01$.

An MDS ordination plot of Bray-Curtis similarity between samples based on production-by-family data (Fig. 3b) demonstrated samples from the High dredging intensity site to be more widely dispersed than those from the Low intensity and Reference sites. This observation was confirmed by the results of a multivariate dispersion test (R=0.705, L=0.830, H=1.465). However, samples from the different sites showed a degree of overlap, suggesting that the assemblages' capacity for Somatic Production had not been permanently altered by dredging. ANOSIM results (Table 3) showed a similar level of production at the Low dredging and the reference site occurring in 2003 (i.e., dredged site not significantly different from the reference sites), seven years after the cessation of dredging. No recovery was apparent at the High dredging intensity site relative to the Reference site.

3.3. Biological Traits Analysis

The species trait-by-sample matrix resulting from Biological Traits Analysis was subject to multivariate analyses. As with previous analysis techniques, the dispersion of samples from the High dredging

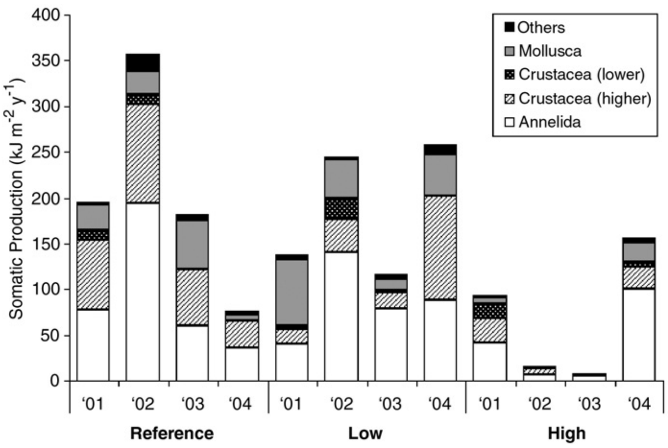

Fig. 4. Relative contribution by major phyla to the total Somatic Production (in kJ m^{-2} y^{-1}) at the High and Low dredging intensity and the Reference sites each year from 2001 to 2004.

intensity site was greater than that of the Low intensity and Reference sites (multivariate dispersion index values: R=0.670, L=0.975, H=1.356), yet there was a high degree of overlap between samples from all sites (Fig. 3c). Again, this suggests that assemblages present at dredged sites can be as functionally diverse as reference samples, as measured by the range of biological traits they accommodate. Samples from dredged sites that were most similar to Reference samples were those furthest in time from when dredging occurred. ANOSIM tests (Table 3) revealed that by 2002 there were no significant differences, in terms of biological traits, between the assemblage from the Low dredging intensity site and Reference sites. In contrast, clear differences in trait composition were evident between the High dredging intensity and Reference sites in all years, particularly during on-going dredging during 2002 and 2003.

SIMPER analysis revealed which traits were responsible for the dissimilarity between Low dredging intensity and Reference sites in 2001, one year before functional recovery was judged to have occurred in 2002. Almost without exception the expression of all trait categories was lower at the Low dredging intensity site in comparison with the reference sites. Whilst this is probably a result of the lower abundances found at this site, SIMPER suggests certain trait categories assume greater importance in discriminating between sites. These categories include ADULT LIFE HABIT_sessile and RELATIVE ADULT MOBILITY_none. Species displaying such traits include members of the subphylum Tunicata and the barnacle *Balanus crenatus*.

3.4. Taxonomic Distinctness

Mean Taxonomic Distinctness (TD) appeared to be reduced slightly by dredging activity (Fig. 2c) but differences between sites at any given time were rarely statistically significant (only High and Low sites differed in 2004). Variation in Taxonomic Distinctness did not differ significantly between sites or over time, and was not affected by resumption of dredging activity at the High intensity site (data not shown).

The extent to which Taxonomic Distinctness of the assemblage at each site differed from expectation (depicted as the mean TD of the whole assemblage (horizontal dotted line) with 95% confidence limits for increasing numbers of species (funnel)) was assessed (Fig. 5). Samples taken from the High dredging intensity site tended to group towards the left of the plot as, on the whole, they contained fewer species than samples from other sites. However, they were not distinctly separated from samples taken from the Low intensity or Reference sites. Neither do any samples appear to group outside the 95% confidence limits of the mean TD for the whole assemblage,

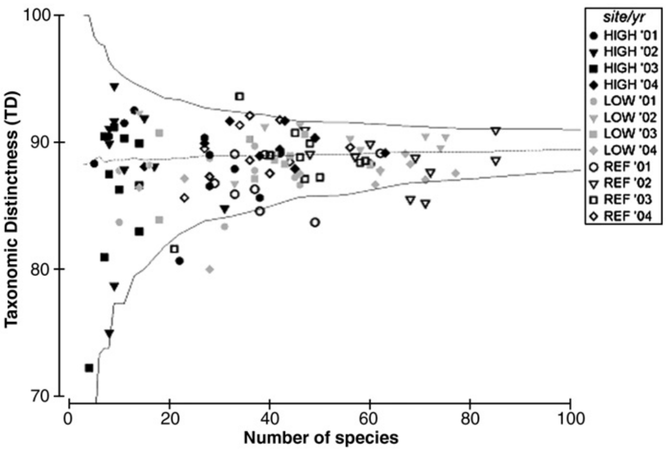

Fig. 5. Funnel plot of Taxonomic Distinctness (TD) of samples from High and Low dredging intensity sites and from Reference sites between 2001 and 2004. The funnel lines indicate the 95% probability limits.

Table 4
Recovery times of the High and Low dredging intensity sites according to each of the functional diversity indices tested

Index	Analysis Technique	Year of Recovery (Number of Years After Dredging)	
		Low Intensity Site	High Intensity Site*
Infaunal Trophic Index (ITI)	Univariate	≤2001 (≤5)	≤2001 (≤5)
Trophic Group Analysis (TGA)	Multivariate	≤2001 (≤5)	>2001 (>5)
Somatic Production (P_S)	Univariate	≤2001 (≤5)	≤2001 (≤5)
Taxonomic Distinctness (TD)	Uni/Multivariate	≤2001 (≤5)	≤2001 (≤5)
Rao's Quadratic Entropy (Q)	Uni/Multivariate	≤2001 (≤5)	>2001 (>5)
Abundance (N)	Univariate	2002 (6)	≤2001 (≤5)
Species Richness (S)	Univariate	2002 (6)	>2001 (>5)
Biological Traits Analysis (BTA)	Multivariate	2002 (6)	>2001 (>5)
Biomass (Ash Free Dry Weight)	Univariate	2003 (7)	≤2001 (≤5)
Multivariate N & S	Multivariate	2003 (7)	>2001 (>5)
Somatic Production (P_S)	Multivariate	2003 (7)	>2001 (>5)

Recovery times based on abundance, species richness and biomass values are from Cooper et al. (2007). * Recovery times of the High dredging intensity site must be treated with caution as dredging resumed during the study.

indicating that the assemblages at the dredged sites were, on average, just as taxonomically distinct as the assemblage at the undisturbed sites. In other words, dredging appeared to have no permanent adverse effect on the Taxonomic Distinctness of the re-colonising assemblage.

3.5. Rao's Quadratic Entropy Coefficient

Rao's Q did not differ between the Low dredging intensity site and the Reference sites, suggesting that the Low intensity site had recovered full ecosystem functionality at least five years after the cessation of dredging (Fig. 2d). At the High intensity site, however, Rao's Q was lower than at the other two sites in 2001 and 2004, but not in 2002 and 2003, during the period of resumed dredging. This seems counter-intuitive, given the known disruptive effects of dredging for aggregate. It is worth explaining this observation here, whilst in context. It has already been mentioned (section 2.2.5) that Rao's Q can behave unexpectedly under certain conditions. It appears that those conditions have been met during the present investigation. Dredging activity has been shown to reduce the numbers of species and individuals in the faunal assemblage at the Hastings Shingle Bank (Cooper et al., 2007). However, the numbers of species and of individuals were not reduced proportionally after disturbance, therefore, the few individuals that remained may have accounted for a greater proportion of the species present, thus elevating the value of Rao's Q. Once the recovery of the site commenced, new arrivals steadily increased the overall species-abundance based diversity of the assemblage but they may have lowered the average dissimilarity among species (as they may not have been adding any new trait to the assemblage). Consequently, functional diversity decreased. It is likely that this process was starting to manifest itself at the High dredging intensity site between 2003 and 2004 as it started to recover from recent dredging activity.

3.6. Index Summary and Comparison

A summary table indicating the recovery times of the High and Low dredging intensity sites according to each of the functional diversity indices tested is presented (Table 4), together with recovery times based on numbers of species, individuals and biomass from Cooper et al. (2007). Since dredging was resumed at the High dredging intensity site in 2002 (six years after the cessation of initial dredging), if recovery was not apparent by 2001, it was deemed to have occurred over a period longer than five years and, therefore, impossible to detect. For this reason, comparisons amongst indices are based solely

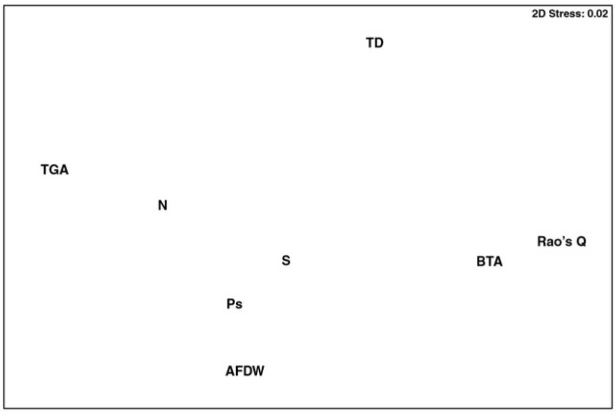

Fig. 6. Inverse MDS plot of Euclidian distance calculated between normalised index-per-sample data. See Table 4 for label definitions.

on the differences observed between the Low dredging intensity site and the Reference sites.

The ITI, TD, Rao's Q and univariate analysis of P_S all indicate that the Low dredging intensity site recovered ecosystem function by 2001, five years or less after cessation of low intensity dredging. This time scale is shorter than that reported by the BTA and univariate analyses of the traditional metrics of numbers of species (S) and individuals (N) (recovery after six years). Biomass and multivariate analyses of P_S and of S and N, all indicate recovery being complete after seven years. Univariate analyses of the ITI, P_S, TD, biomass and number of individuals suggest that functional recovery also occurred by 2001 at the High dredging intensity site (i.e., before the resumption of dredging).

Finally, an inverse MDS ordination plot of Euclidian distance calculated between normalised index-per-sample values (Fig. 6) illustrates the relative similarity between different indices. For instance, BTA and Rao's Q both rely on multiple trait information and, therefore, appear relatively close to one another. Similarly, P_S and AFDW (biomass) rely on species' weight and on single species-specific conversion factors; they also appear close together. The variation between these pairs of indices may be related to their use of multiple (BTA, Rao's Q) versus single traits (P_S, AFDW) in their determination. TD, unlike any other index, is based on species' relatedness and appears distinct from all other indices. ITI uses a severely reductionist approach to represent an assemblage (in this case, converting the information contained within 353 species and their relative abundance into just four categories) and also appears distant from most other indices. The central placement of S reflects that it is almost equidistant from all indices and, indeed, all indices rely on S for their calculation. N provides information that falls somewhere between that given by S and by the ITI.

4. Discussion

The results presented above represent an important step towards attaining a better understanding of the effects of aggregate extraction on benthic macrofaunal assemblages. They also serve to demonstrate how alternative indices behave under changing environmental conditions, relative to traditional metrics.

It appears that indices for measuring ecosystem function are as diverse as the assemblages they are designed to characterise, yet it seems reassuring that those tested during the present investigation all show some degree of congruence, despite their differing methods of calculation. Where reliable comparisons between indices have been possible (i.e., between their values at the Low dredging intensity site and Reference sites), recovery times have differed only by one or two years between indices (Table 4). Considering the different methods by which each index is calculated, this work gives some confidence that each index reflects, at least in part, what is happening to the macrofaunal assemblage and its functional contribution.

From the perspective of this investigation, it is unfortunate that dredging activity resumed at the High intensity site, as it was hoped that the difference in treatments between sites would have added valuable information about potentially varying rates of recovery. The beneficial aspect of the resumption of dredging at the High intensity site is that the immediate effects of such activity on the benthos can be studied in synchrony, and the behaviour of each index tested under extreme disturbance conditions. Again, this exercise could not have been more apt given the intuitively unexpected behaviour of Rao's Q (which, unlike any other index, appeared to indicate an increase in functional diversity during periods of active dredging). Furthermore, it also demonstrated the importance of investigating short-term recovery immediately after dredging. However, the downward fluctuation in macrofaunal abundance and diversity at the reference sites in 2004 meant that, in comparison, impacted assemblages appeared statistically indistinguishable in this year. Subsequent data would be required in order to assess the short-term recovery at the high dredging intensity site.

Despite the unexpected complications arising during the investigation, there remains little doubt that ecosystem function can recover fully, in time, at the Hastings Shingle Bank, as determined using the proxy indices for functionality employed. At the Low dredging intensity site, which was subjected to less than 1 hour of dredging per 100 m^2 in the year 1996 (Cooper et al., 2007), the functional capacity of the macrofaunal assemblage appeared to have recovered one or two (perhaps more) years before recovery as measured by traditional techniques (depending on which ecosystem function index is used). The Infaunal Trophic Index, Rao's Q, Taxonomic Distinctness and univariate analysis of Somatic Production all suggested that recovery had already taken place by the start of this investigation, at least one year before the time suggested by traditional techniques (Table 4). At the High dredging intensity site, which was subjected to more than 5 hours of dredging per 100 m^2 in the year 1996 (Cooper et al., 2007), recovery was also apparent by 2001 according to Taxonomic Distinctness and univariate analyses of ITI and P_S. However, this time scale for recovery was also apparent from traditional measures of abundance and biomass at this site.

Multivariate analysis of the various index values calculated for the present investigation confirmed that assemblages sampled at dredged sites could be functionally as similar as those from Reference sites (given the degree of overlap in sample scatter illustrated in all MDS plots). Samples most dissimilar to Reference ones were those collected closest to the time of disturbance (e.g., highly scattered samples on MDS plots were likely to be those collected at the High dredging intensity site during 2002 and 2003). Yet, as time progressed, samples from impacted sites slowly became more similar to reference samples. As a result of the tight coupling between benthic species and sediment variables, this suggests that the physical nature of the seabed was not permanently altered after dredging, as it was able to accommodate, in time, the same assemblage as existed before disturbance (allowing for natural variations in population dynamics). A hypothetical alternative scenario where, given time, assemblages do not return to being similar to those under reference conditions may indicate that the seabed has been physically altered to the extent it can no longer accommodate its original inhabitants. Whether this scenario would be detrimental to ecosystem function remains to be investigated.

Naturally, ecosystem recovery may mean different things to different people, therefore it would be unwise to recommend any one of the indices tested over any other for every given situation (especially as each index uses different information for its calculation and provides its own unique interpretation – Fig. 6). For example, a conservationist may be interested in the preservation and recovery of as taxonomically diverse an assemblage as possible, in which case, assessment of the potential for recovery using Taxonomic Distinctness may be desirable. Alternatively, a commercial fisherman may be interested to find out if and when there has been a full recovery of the benthic food source for his targeted fish species, in which case Somatic Production may be most appropriate for his needs. In cases where a disturbance permanently alters the physical or chemical nature of the benthic environment, the Infaunal Trophic Index could be used to assess functional recovery given its reliance on Pearson and Rosenberg's (1978) model for organic enrichment, and even when monitoring habitat restoration experiments, Rao's Q or Biological Traits Analysis may be most suitable, depending on the aims of such initiatives. The present investigation has served to compare different methods of assessing ecosystem function and each one has, on the whole, proved to be a useful tool. All indices tested are seen as complementary to the traditional metrics of assessment. It remains to be seen how each behaves under conditions where dredging activities permanently alter the physical nature of the seabed, whether by changing the nature of patches though practises such as sediment screening or by homogenisation of formerly varied regions (Thrush et al., 2006).

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Paper #9

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Long-term benthic responses to sustained disturbance by aggregate extraction in an area off the east coast of the United Kingdom

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ABSTRACT

The temporal variability of benthic assemblages inhabiting offshore gravel deposits in the North Sea is poorly known, as purpose-collected long-term data sets have until recently been non-existent. It has therefore proved difficult to understand the stability and resilience of these benthic ecosystems after disturbance caused by the extraction of aggregates on an industrial scale. The present investigation examines an 8-year time series of data collected in and around an active commercial aggregate extraction site off the east coast of the United Kingdom. Both physical and biological data sets suggest a distinct yet localised effect after sustained gravel extraction, with impacted sediments generally appearing more physically homogeneous and faunistically impoverished than undisturbed sediments. Although inter-annual variability of selected assemblage metrics was reduced in disturbed sediments, differences in some assemblage metrics became significant between years. Despite such observations, significant impacts to the benthos in any given year were not sustained for long. However, the magnitude of impact in almost every year would be enough to merit remedial intervention based on an existing model of measuring acceptable levels of disturbance as a result of organic enrichment. Caution must be exercised in making any such recommendations, especially as there are presently no models specifically designed to assess the degree of acceptable disturbance from aggregate extraction. This study not only highlights the importance of and need for long-term data sets in order to better understand the difference between natural and human-induced variability in benthic assemblages, but also emphasises the need to develop more relevant monitoring tools to better manage the activities of the marine aggregate extraction industry.

1. Introduction

Each year, sand and gravel are dredged from the seabed around the UK as a source of aggregate for the construction industry, to supplement land-based sources, or as a source of material for beach nourishment and coastal defence (Singleton, 2001). The total contribution of marine aggregate to UK supplies between 1955 and 2002 has been around 500 million tonnes and marine aggregate makes up around 21% of the current production in England and Wales. Yearly landings from licensed extraction areas around England and Wales are regularly more than 20 million tonnes (Gubbay, 2003, 2005).

The localised distribution of suitable marine aggregate deposits has led to intense extraction activity in specific localities, particularly around the Isle of Wight in the central English Channel and in an area off Great Yarmouth known as Cross Sands, in the southern North Sea. The number of extraction licenses in both these areas has, in recent years, led to concerns regarding the potential for cumulative short and long-term environmental impacts, particularly as extraction activities often overlap with those of other stakeholders such as the fishing industry (Cooper et al., 2007).

To date, there has been a substantial research effort directed at understanding and mitigating potential impacts of marine aggregate extraction (e.g., de Groot, 1996; Kenny and Rees, 1996; Seiderer and Newell, 1999; Boyd and Rees, 2003; Boyd et al., 2005). However, existing knowledge on the cumulative impacts of intensive dredging is more limited (Desprez, 2000; Cooper et al., 2007). Cumulative impacts have been defined as "effects on the environment, either from the summation of individually minor but collectively significant impacts, or as a result of the interaction of impacts from one or more sources" (DTLR, 2002). Surveys designed to assess the potential for cumulative environmental impacts arising from aggregate extraction, therefore, need to consider the sum of individual impacts from individual existing licensed extraction sites, both in space and time (including changes projected into the foreseeable future). Any attempt at understanding the cumulative effects of marine aggregate extraction requires the conduct of new, carefully targeted sampling regimes to cover appropriate spatial scales (e.g., Cooper

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et al., 2007) and to establish the stability of any observed effects over time. This temporal (year-on-year) element to sampling is critical to establish with confidence trends in the data and to determine whether observed patterns in the distribution of resources are simply an artefact of sampling on one occasion only. Factors such as annual variability in dredging intensity and natural variability in benthic populations mean that 'one-off' sampling can hinder the effective evaluation of the potential for cumulative impacts of marine aggregate extraction. It is the aim of the present study to investigate the potential for cumulative environmental effects arising from sustained marine aggregate extraction at Cross Sands using time-series data, as well as to examine and compare the temporal variability in disturbed versus undisturbed benthic assemblages. The sequential data set analysed in this investigation represents the first and only long-term (>5 years) data available from offshore gravel deposits in the North Sea. Research on cumulative effects is a major departure from conventional 'once off' evaluations of the impact of dredging in that it aims to evaluate the interaction of events separated in time and in space.

2. Methods

2.1. Study site

The industrial dredging site of Cross Sands comprised 15 adjacent licensed areas covering an approximate area of 340 km^2 located between 5 km and 25 km offshore along the east coast of the United Kingdom (Fig. 1). The area spans water depths from 15 m to 41 m below chart datum. The underwater environment at Cross Sands is a highly dynamic system with strong tidal currents and mobile bedforms. The sediment deposits within the area consist mainly of sandy gravels and gravelly sands.

Cross Sands was first licensed for marine sand and gravel extraction in 1969. Between 1975 and 1989 a total of 148,255,194 tonnes of material was extracted. From 1989 to 2005 annual extraction levels stabilised at around 9 million tonnes, peaking in 1990 at 11 million tonnes. The Cross Sands location remains the most intensively dredged accumulation of marine aggregate extraction licensed sites in the UK (The Crown Estate and BMAPA, 2005).

There is limited evidence of benthic sampling in this locality prior to the conduct of this study. However, those studies that have been carried out have described the benthic macrofaunal assemblage as being impoverished and characterised by the polychaete *Ophelia borealis* (Kenny et al., 1991; Cooper et al., 2007). The limited range and density of macrofauna in this area has been attributed to

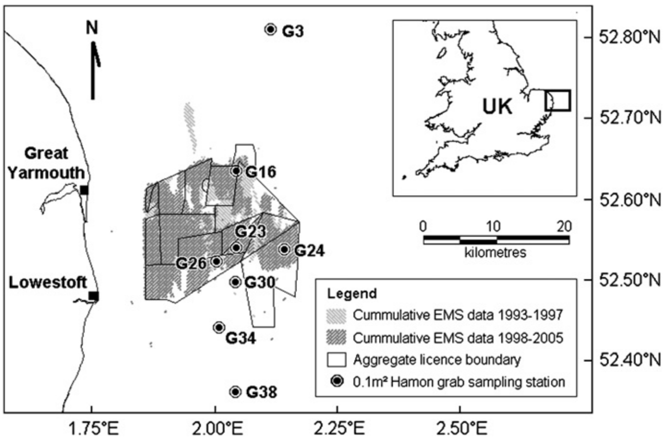

Fig. 1. Location of sampling stations in relation to the distribution of aggregate extraction licences at Cross Sands. Shaded areas represent the cumulative footprint of direct aggregate extraction (based on annual EMS records) over the periods of 1993–1997 and of 1998–2005.

the abrasive effects of shifting sands under strong tidal currents and storm action. Locally, there are also biogenic reefs of the gregarious polychaete *Sabellaria spinulosa*, which act to stabilise mobile sediments by utilising sand in the construction of their tubes. The resulting more stable substrate permits a greater range and diversity of species to become established (Cooper et al., 2007).

2.2. Survey design and sampling strategy

In 1998, Cooper et al. (2007) sampled a grid of 39 stations in the vicinity of the Cross Sands extraction licensed areas. Their sampling regime was designed to provide a snapshot of broad-scale patterns in the macrobenthic assemblage in the region. A sub-set of 8 sampling stations were selected from their grid survey to represent a near linear transect through the region. Subsequent annual sampling of the sub-set of stations was intended to contribute towards a time-series data set to assess the persistence of observed physical effects across the region and to investigate temporal variation in macrobenthic assemblage structure in relation to changes in dredging intensity. The 8 stations selected were (listed from North to South) G3, G16, G23, G24, G26, G30, G34 and G38 (Fig. 1). Stations G16, G23, G24 and G26 fell inside the boundaries of the licensed extraction area.

Between 1998 and 2005, annual surveys (in June) collected up to four sediment samples at each station using a 0.1 m^2 Hamon grab deployed from Cefas' research vessel. Stations were located using a differential Global Positioning System and the ship's software that logs the position of sampling. Once on deck, the total volume of each grab sample was measured and a sub-sample of sediment (approx. 500 ml) was removed for particle size analysis. The remaining sediment was washed over 1 mm mesh sieves and the retained residue containing the benthic macrofauna was fixed in a 4% formaldehyde solution for later processing in the laboratory. Over the years, accidental damage of a very limited number of samples has resulted in unequal numbers of samples being available for ecological and particle size analyses.

2.3. Sample processing

Two hundred and four macrofaunal samples were used for ecological analyses. Macrofauna was extracted from the samples by viewing under light magnification and preserved in 70% Industrial Methylated Spirit. All animals were identified to the lowest possible taxonomic level and each taxon enumerated and weighed, after blotting, to the nearest 0.0001 g. Ash-free dry weight (biomass) was calculated using conversion factors in Rumohr et al. (1987) and Ricciardi and Bourget (1998). Colonial species were recorded as present. A representative reference collection of collected specimens was sent for external taxonomic verification.

Particle size analyses were conducted on sediment sub-samples taken from 212 grab samples. Sediment samples were initially wet sieved on a 500 μm stainless steel test sieve, using a sieve shaker. The <500 μm fraction was freeze-dried, weighed and a sub-sample analysed using a Coulter LS 130 Laser-Sizer. The >500 μm fraction was oven dried at 80 °C for 12 h and sieved over a range of test sieves down to 500 μm at 0.5 phi intervals. The sediment retained on each sieve was weighed to he nearest 0.01 g. The results from these analyses were combined to give a full particle size distribution. Sediment type descriptions were based on the Wentworth scale (Bale and Kenny, 2005).

2.4. Data analysis

Since 1993, all marine aggregate dredging vessels working on UK licences have been fitted with Electronic Monitoring Systems (EMS) that automatically record a dredger's position and its dredging status every 5 s. The Crown Estate, who manage the UK marine aggregate resource, use EMS data and a block analysis technique to produce maps showing the location and intensity of dredging for all licensed extraction areas. These maps show the seabed divided into 50 m × 50 m cells, each cell coloured according to dredging intensity in any one year. Such maps were obtained for each year between 1993 and 2005 in the form of raster images. Using the Geographic Information System software package Mapinfo®, these images were geo-referenced and a boundary drawn around all areas directly impacted by dredging to create a dredging footprint for each year. Yearly records were combined to show the cumulative area dredged over various time intervals. Maximum dredging intensity (in hours per year) was also noted at the precise cell in which each of the impacted sampling stations fell. A degree of caution must be observed when interpreting EMS maps and data, as dredging intensity values are likely to be overestimates. This is because a 50 m × 50 m cell can register as being dredged even when it has only been 'touched' once by the 2 m wide draghead.

Univariate analytical techniques were performed on raw data consisting of the number of individuals (i.e., abundance *A* – excluding colonial species) and the number of taxa (hereafter referred to as species richness *S*). Hill's (1973) diversity and evenness indices, *N1* and *N2* respectively, were calculated for each sample. Differences in mean values for each of the above variables between different sampling stations, years and treatments (i.e., inside vs. outside licensed extraction area) were tested for significance using a General Linear Model ANOVA, followed by pairwise comparisons using the Tukey–Kramer method to examine which pairs of means were different. These analyses were performed using the Minitab® software package, version 15.

Rees and Pearson (1992) developed an index, together with guidance/limit values, to test for the acceptability of change at stations within the sphere of influence of a disturbance relative to reference conditions. Their calculated guidance values were based on Pearson and Rosenberg's (1978) empirical model for the responses of benthic communities to organic enrichment, and have subsequently been applied by Rees et al. (2006a,b) both to a long-term data set investigating the effect of sewage-sludge disposal, and to a shorter data set arising from a study of a relinquished marine aggregate extraction site. Data acquired for the present study has been used to further illustrate the use of this approach and test its utility in assessing the impacts of continuing marine aggregate extraction. Calculated index values have been compared with the same guidance values as those defined by Rees and Pearson (1992) despite the nature of the disturbance presently under investigation being one of extraction rather than addition of material. Although it is recognised that guidelines based on disposal/enrichment regimes may not be ideally suited to evaluating the specific effects of dredging on the benthos, there is evidence to suggest that the outwash from marine aggregate dredging may provide some organic enrichment in the vicinity of dredging operations (Newell et al., 1999). It was deemed worthwhile, therefore, to determine the suitability of existing guidance values for assessing the impacts of sustained aggregate extraction. Guidance values aside, the power of the test lies in its ability to detect significant departures of the index from equity (i.e., zero), which would denote a significant effect of the disturbance on the benthic assemblage relative to reference conditions. Testing this involved pairwise comparisons of quantitative measures (i.e., abundance and species richness values), as follows:

$$[(\text{Treatment}/\text{Reference}) - 1] \times 100$$

Means with 95% bootstrapped confidence intervals for pairwise comparisons of annual measures were calculated using the R© software package, version 2.4.1 (R Development Core Team, 2006).

The Primer® package, version 6 (Clarke and Warwick, 1994), was used for multivariate analyses. The zero-adjusted Bray-Curtis similarity coefficient (Bray and Curtis, 1957; Clarke et al., 2006), which is adjusted for sparse samples, was calculated between groups of samples from each sampling station after fourth-root transformation of the raw data. To ease interpretation of the resulting similarity matrices, ordination routines were performed, producing multidimensional scaling (MDS) plots. Analysis of similarities (ANOSIM) was used to test the significance of the difference in similarity between various groups of samples defined prior to analyses (e.g., between years, stations and treatments). The similarity percentages (SIMPER) routine was used to establish which species contributed the most to the observed differences in the data. The significance of the relationship between similarity matrices obtained from environmental and biological data was tested using the RELATE routine. Finally, the BIOENV routine was used to identify the environmental variables that best explained the patterns observed in the biological data. This was achieved by selecting sub-sets of environmental variables that maximised the rank correlations between the two matrices. Measured environmental variables included % gravel, % coarse sand, % medium sand, % fine sand, % silt/clay, sorting, mean water depth (m) and bed stress ($N\,m^{-2}$).

3. Results

3.1. Physical impact

The spatial extent of dredging for marine aggregates in and around the Cross Sands licensed extraction area between 1993 and 2005 is illustrated in Fig. 1. The cumulative temporal extent of dredging is presented in Table 1. It would appear that over this 13-year period, approximately three quarters of the licensed area has been impacted directly by dredging activities. Of the four sampling stations within the licensed area, G23 has been dredged most intensively (up to 55 h, Table 1), followed by G24, G16 and G26. Since 1998, all stations except for G23 have been dredged for little over 1 h. G23 has received up to 26 h of direct dredging over the same period.

The effect of the dredging regime on the sediments sampled seems apparent from the mean sorting coefficients for each sampling station (Table 1). On the whole, stations falling within the licensed extraction area (G16, G23, G24 and G26) have a significantly lower mean sorting coefficient than those falling outside the licensed area (ANOVA *F* value: 47.86, $p = 0.000$). A lower sorting value characterises a more homogeneous sample, whereas higher sorting values are indicative of a more varied mix of particle sizes. It would appear that stations that have undergone sustained dredging and selective screening of gravel particles have acquired a relatively more uniform physical character. Obviously, without any baseline data on the nature of the sediments before aggregate extraction commenced, it is impossible to confirm that this observation is solely due to dredging activities. Medium sand was the predominant particle size class at most sampling stations. The northernmost station (G3) was characterised by a greater proportion of fine sand, whereas southernmost stations (G30, G34 and G38) had a higher proportion of gravel than other stations. An illustration of which physical parameters best differentiate each sampling station can be seen in Fig. 2.

3.2. Biological impact

Over the 8-year sampling period, a total of 13,318 individuals was enumerated, belonging to 250 taxa. An additional 38 taxa were classified as colonial and not incorporated into ecological analyses. Overall, polychaetes were the numerically dominant component of

Table 1
Selected biological and physical parameters of each sampling station at Cross Sands. Mean values are calculated per replicate sample (0.1 m^2). Highest mean values for each parameter evaluated appear in bold text. Cumulative dredging intensity values represent upper limits reported by The Crown Estate and BMAPA annual reports 1993–2005 (The Crown Estate and BMAPA, 2005)

	G3	G16	G23	G24	G26	G30	G34	G38
Macrofauna								
Number of replicates *n*	23	21	29	23	22	29	28	29
Mean abundance *A*	**379.0**	6.8	9.1	14.9	38.6	37.8	46.2	21.3
Mean species richness *S*	**30.2**	3.8	5.3	7.1	8.0	11.4	15.1	7.6
Mean diversity *N1*	**11.5**	3.1	4.1	5.3	4.8	6.5	6.9	4.9
Mean evenness *N2*	**7.4**	2.9	3.7	4.5	3.9	4.8	4.9	4.0
Mean biomass AFDW (g)	**0.467**	0.033	0.006	0.029	0.034	0.039	0.083	0.034
Particle size analysis								
Number of replicates *n*	26	21	29	26	25	28	28	29
Mean % gravel	10	11	14	20	12	28	29	29
Mean % coarse sand	20	32	**48**	17	19	22	13	18
Mean % medium sand	21	**44**	33	**53**	**52**	**36**	**30**	**35**
Mean % fine sand	**35**	11	5	9	13	12	18	8
Mean % mud/silt clay	13	2	0	1	4	2	10	10
Mean sorting	2.3	1.4	1.4	1.7	1.6	2.3	2.1	2.4
Cumulative dredging intensity (h)								
1993–1997	–	7.0	28.5	10.0	5.0	–	–	–
1998–2005	–	1.0	26.2	1.0	1.3	–	–	–
1993–2005	–	8.0	54.7	11.0	6.2	–	–	–

the macrofauna, comprising 53% of the total assemblage. They were followed by molluscs (17%), crustaceans (15%), 'others' (9%) and echinoderms (6%). The cumulative effect of dredging for aggregates had no significant effect on the proportion of each phylum in the macrofaunal assemblage inside the licensed area relative to outside.

Fig. 3 illustrates the difference in variability of selected assemblage parameters between sampling stations. Station G3 displayed the greatest variability for all assemblage parameters over the 8-year sampling period. G3 was also unique in having significantly higher values for all assemblage parameters than any other station or groups of stations. This is most likely due to the presence at G3 of *Sabellaria spinulosa*, a reef-building polychaete known to enhance the physical complexity of sandy substrates, thus providing additional structures for other benthic species to inhabit (Connor et al., 1996).

Year-to-year, each station displayed significant differences in at least one of the five assemblage parameters evaluated (Table 2). No individual station displayed significantly different values for all assemblage parameters, and no single parameter was consistently different between years at all sampling stations. Species richness (*S*), diversity (*N1*) and evenness (*N2*) of the macrofaunal assemblage within the licensed aggregate extraction area differed significantly between years. Outside the licensed extraction area, no significant differences were observed in any assemblage parameter between sampling years.

Multivariate analyses revealed that assemblages were significantly dissimilar both between sampling stations (ANOSIM Global *R*: 0.582, 0.1% significance) and between sampling years (ANOSIM Global *R*: 0.396, 0.1% significance). An MDS plot of all samples taken over the entire sampling period also reveals a tighter clustering of replicates taken within the licensed aggregate extraction area than those taken outside (Fig. 4). Although samples taken from inside and outside the licensed extraction area overlap on the MDS plot, statistical analysis proves that the assemblages in each treatment category are significantly dissimilar (ANOSIM Global *R*: 0.195, 0.1% significance. NB: exclusion of G3 samples from the analysis did not alter the significance of the ANOSIM result). The SIMPER routine revealed that the average similarity amongst samples falling inside the licensed aggregate extraction area was higher (23.12) than that amongst samples outside the area (17.26). The average dissimilarity between both groups of samples was 85.32. In addition, inside the licensed area only eight species contributed towards 90% of the similarity within that group of samples, whereas 23 species contributed towards 90% of the similarity amongst samples outside the licensed area. Conducting the same multivariate analyses using biomass data reveals almost identical results (data not shown).

3.3. Temporal variability

When temporal variability in assemblage similarity is considered for each sampling station individually, all stations except for G16 appear to have significantly dissimilar assemblages between sampling years (according to the significance of the *R* statistic obtained after ANOSIM – Table 3). G16 appears to be unique in retaining a statistically similar macrofaunal assemblage from year to year.

The BIOENV routine was performed to identify which environmental variable or combination of variables could best explain the difference in similarity between assemblages at each station over all sampling years. The global sample statistic (Spearman's Rho), together with its significance level, is displayed in Table 3 for each

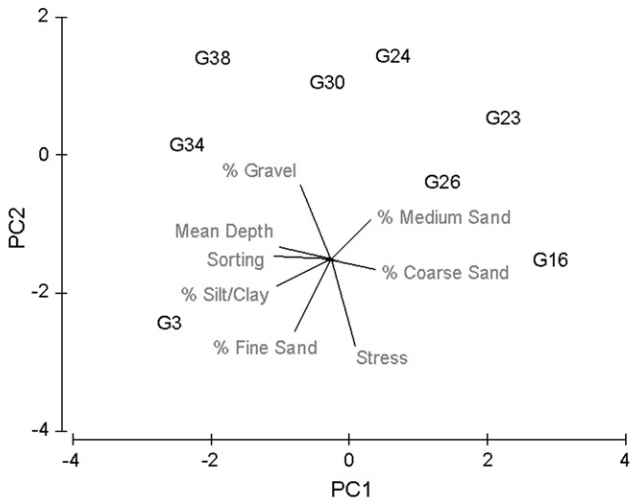

Fig. 2. Principal Components Analysis of measured physical parameters at all sampling stations (averaged over eight years).

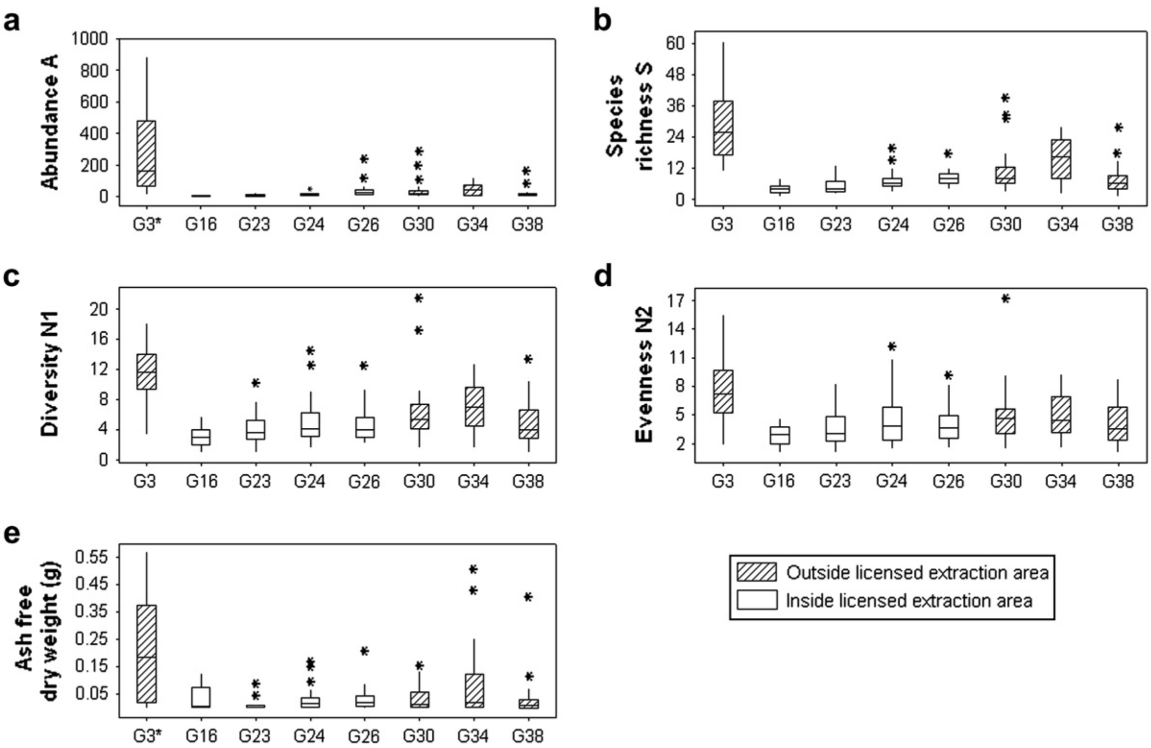

Fig. 3. Boxplots illustrating the range in values of selected benthic macrofaunal assemblage parameters recorded per 0.1 m^2 at each sampling station. Measured parameters include abundance (*A*), species richness (*S*), Hill's diversity (*N1*) and evenness (*N2*) indices, and biomass (ash-free dry weigh in g). Hatched boxes represent stations falling outside the licensed aggregate extraction area. * The ordinate axes in 3a and 3e have been truncated, thereby excluding outlier values of 2405 individuals and 4.42 g, respectively, recorded at G3.

sampling station. Only at stations G3, G24, G26, G30 and G34 did any measured environmental variable explain the difference in similarity between assemblages over the years (i.e., the significance level of Rho was below 5%). However, no individual environmental variable exerted a significant influence in isolation. Instead, the significant scores were caused by a combination of variables, most notably those describing the particle size distribution of the sediment. Only at G26 did mean depth play any role in shaping the macrofaunal assemblage to a significant degree.

Since Spearman's Rho and the ANOSIM *R* statistic are not directly comparable, it is not straightforward to ascertain whether the effect of temporal variability is larger or smaller than the influence of any measured environmental variable on the assemblage within a sampling station. However, there is a way of producing a Spearman Rho value that corresponds to the ANOSIM *R* by performing a RELATE test comparing the biotic similarity matrix against a model matrix created from unordered groups, where the group structure is the different years (i.e., with 0's in years that are the same and 1's in years that are different; Clarke, personal communication). Results from such a RELATE test are presented in Table 3, and at all stations except G23, the RELATE Rho is smaller than the BIOENV Rho. It would appear, therefore, that only at G23 is temporal variability of greater importance in shaping the assemblage than the measured environmental variables. In other words, once the provenance of a particular sample is known, and secondarily the year in which it was taken, there is no further information about the sampled assemblage that can be gleaned from individual environmental variables. It is worth recalling that station G23 has been subjected to the greatest dredging intensity of any station within the licensed extraction area (up to 54.7 h spread evenly over all years since 1993; Table 1); therefore, the detection of assemblage dissimilarity between years at this station is not surprising.

An important element of the present investigation is the employment of environmentally comparable treatment and reference

Table 2
F values and significance (*P*) values resulting from ANOVA tests on selected macrofaunal assemblage parameters calculated for each sampling station and treatment (i.e., whether inside or outside the licensed aggregate extraction area). Assemblage parameters were compared between years and included abundance (*A*), species richness (*S*), diversity (*N1*), evenness (*N2*) and ash free dry weight (AFDW). Statistically significant *P* values ($\leq$5%) are displayed in bold text

	n	*A*		*S*		*N1*		*N2*		AFDW	
		F	*P*	*F*	*P*	*F*	*P*	*F*	*P*	*F*	*P*
G3	23	9.69	**0.000**	1.79	0.162	0.87	0.551	1.41	0.271	0.53	0.802
G16	21	1.82	0.167	2.30	0.093	2.05	0.126	1.82	0.167	4.37	**0.011**
G23	29	4.15	**0.005**	7.30	**0.000**	5.95	**0.001**	4.10	**0.005**	1.22	0.333
G24	23	1.99	0.125	2.84	**0.042**	3.13	**0.030**	3.75	**0.015**	1.61	0.207
G26	22	1.06	0.436	2.85	**0.045**	4.57	**0.008**	5.21	**0.004**	0.91	0.524
G30	29	1.11	0.395	1.04	0.432	0.84	0.570	0.99	0.466	2.84	**0.030**
G34	28	4.24	**0.005**	2.23	0.075	0.93	0.503	0.72	0.655	7.37	**0.000**
G38	29	6.88	**0.000**	6.55	**0.000**	1.44	0.243	0.75	0.362	0.84	0.570
In	95	0.99	0.445	5.07	**0.000**	6.80	**0.000**	6.93	**0.000**	1.92	0.073
Out	109	0.73	0.648	0.79	0.595	0.76	0.620	0.48	0.848	0.86	0.540

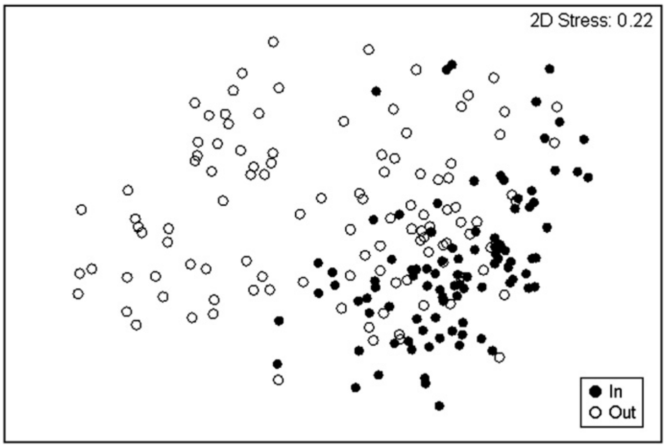

Fig. 4. MDS plot of all macrofaunal samples collected over 8 years coded by treatment (i.e., whether they fall inside or outside the licensed aggregate extraction area).

stations (i.e., stations falling inside and outside the licensed aggregate extraction area, respectively). In both circumstances, responses to natural changes over time may be expected to be in synchrony, allowing ratios of measures at each to be evaluated solely from the standpoint of the treatment of interest, in this case the disturbance caused by the extraction of aggregate. Evidence of synchronicity in selected variables between treatment (In) and reference (Out) stations over the 8-year sampling period is illustrated in Fig. 5a–c, and is especially supported by the significant correlation between values for total species richness ($r = 0.786$; d.f. = 6; $p = 0.02$ – Fig. 5b).

Plots of ratios of abundance (A), species richness (S) and the A/S index between treatment (T) and reference (R) stations indicate that the dredging activity has a varied yet not entirely unpredictable effect on the macrofaunal assemblage (Fig. 5d–f). Ratio values significantly lower than 0 indicate marginal impoverishment throughout the dredging period. Such instances occurred irregularly (in 1999, 2003 and 2005) and were not sustained over periods of more than one year. On the whole, despite a high degree of variability, the effect of dredging during most years of the present study rarely departed significantly from an equitable state. This is not to say, however, that the effects of dredging remained within the limits of acceptable change (delimited by dotted lines) drawn from the empirical model of Pearson and Rosenberg (1978) for the responses of benthic communities to organic enrichment. In this respect, given the degree of disparity beyond the acceptable limits, the effects of dredging activity at Cross Sands could trigger a follow-up investigation. Clearly, further work is required in order to devise similar thresholds modelled on the effects of marine aggregate extraction.

Table 3
Output statistics with significance levels (%) from ANOSIM, BIOENV and RELATE routines testing for differences in similarity between years at each sampling station. Refer to main text for rationale behind each test. Superscript abbreviations refer to environmental variables: Gr = % gravel; CS = % coarse sand; MS = % medium sand; FS = % fine sand; S/C = % silt/clay D = mean depth

	ANOSIM		BIOENV		RELATE (Model)	
	R	Significance (%)	Rho	Significance (%)	Rho	Significance (%)
G3	0.509	0.1	0.387	$4.0^{CS, FS, S/C}$	0.272	0.1
G16	0.070	21.5	0.307	8.0	0.038	20.2
G23	0.497	0.1	0.161	28.0	0.254	0.1
G24	0.392	0.1	0.487	$1.0^{CS, MS, FS, S/C}$	0.203	0.1
G26	0.480	0.1	0.447	$1.0^{MS, D}$	0.254	0.1
G30	0.405	0.1	0.436	$3.0^{MS, FS}$	0.206	0.1
G34	0.442	0.1	0.705	$1.0^{Gr, S/C}$	0.224	0.1
G38	0.285	0.2	0.162	24.0	0.145	0.1

4. Discussion

The composition and structure of the observed benthic macrofaunal assemblage at Cross Sands is typical of high-energy sedimentary environments in the southern North Sea. Numerically dominant species (*Sabellaria spinulosa*, *Abra alba*, *Pisidia longicornis*, *Lanice conchilega*, *O. borealis*), and those caught most frequently (*Ophelia borealis*, *Nephtys cirrosa*, *Polycirrus medusa*, *Ophiura* spp., Nemertea, *Scoloplos armiger*, *Spiophanes bombyx*), are known to be associated with sandy sediments, and are characterised as ‘r-selected’ species with short generation times, rapid reproduction and high dispersal potential. The total number of taxa identified in this study (288) lies at the top end of the range (83–289) reported by other researchers working in the area (Desroy et al., 2002; Boyd et al., 2003, 2005; Van Hoey et al., 2004; Cooper et al., 2007). Given the high sampling effort undertaken by this study, it is unlikely that many more macrofaunal species will be encountered with further sampling at Cross Sands.

Multivariate analyses revealed that each sampling station had a distinctive macrofaunal assemblage. Localised variations in the proportion of the different particle size components of the sediment were likely to be responsible for such differences in faunal similarity (Cabioch, 1968; Kenny et al., 1991; Snelgrove and Butman, 1994). Spatial variability in sediment composition was, in turn, likely caused by medium-scale (100–1000 m) differences in hydrological conditions within the area, which influences the position and dynamics of mobile bedforms such as sand waves (Warwick and Uncles, 1980; Holme and Wilson, 1985). The most abundant, diverse and variable faunal assemblage detected at G3 was found to contain patchy *Sabellaria spinulosa* reefs. Colonies of *S. spinulosa* are known to build tubular reefs out of sand and, befittingly, G3 was characterised by having a relatively high proportion of fine sand (Fig. 2). Interestingly, some samples at G3 could be as faunistically poor as those from the poorest stations encountered in this study (e.g., G16 and G23). Therefore, it may only be the ephemeral and sporadic presence of *S. spinulosa* that encourages the settlement of other organisms in an otherwise sparsely populated benthic environment. As at all other sampling stations, within-station faunal variability most probably can be attributed to fluctuations in population dynamics, which are in turn affected by a broad range of factors from climate to competition (Constable, 1999).

The broad-scale pattern in faunal abundance and species richness values detected along the north-south transect in this study (i.e., relatively high values to the north of the study area, followed by relatively low values in the middle and ending with intermediate values at the south) largely corresponds to the pattern detected by Cooper et al. (2007) in their extensive 1998 sampling snapshot. This would suggest that despite the small-scale (within-station) variability in assemblage structure, broad-scale (between-station) patterns are persistent over the years. Despite a lack of baseline data collected before dredging commenced in the area, it is likely that the impoverished assemblage observed towards the middle of the transect is a direct result of sustained aggregate extraction activities.

4.1. Cumulative effects of dredging on the benthos

Screening of sediment onboard a dredger increases the concentration of the desired gravel fraction by discarding the unwanted sandy fraction overboard. The associated sediment plumes that descend from active dredgers have been shown to settle within 300–500 m downstream from the site of dredging (Whiteside et al., 1995; Hitchcock and Drucker, 1996; Newell et al., 1998). Prolonged

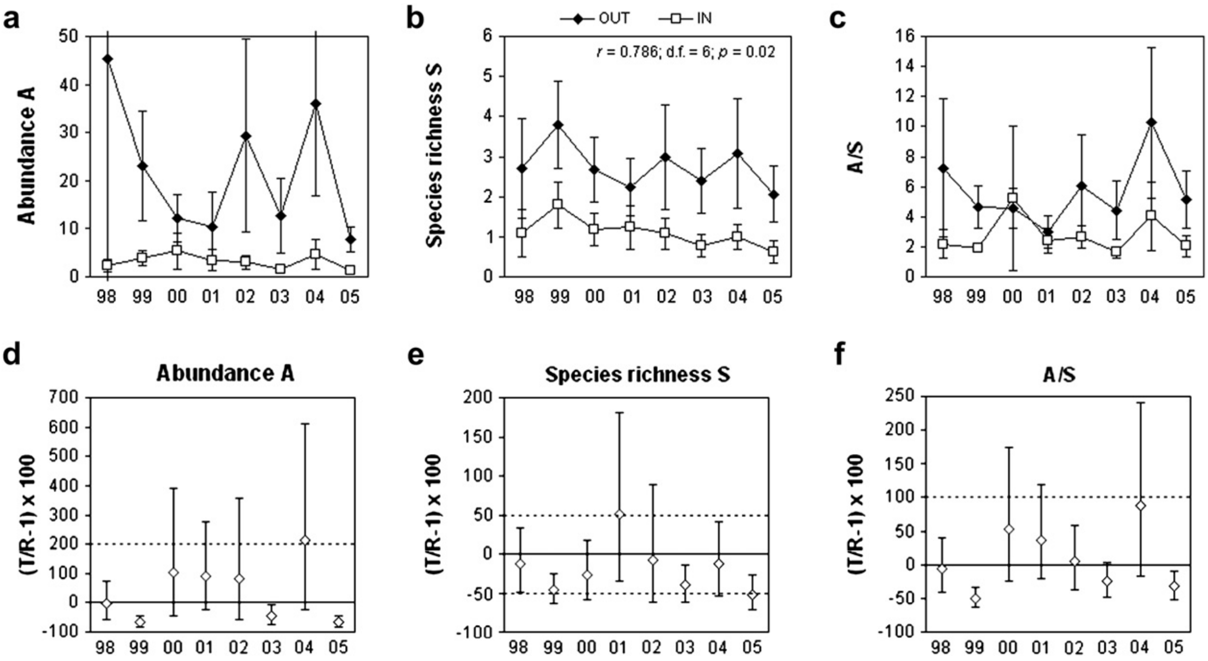

Fig. 5. Temporal trends in sample means with 95% confidence intervals of (a) the abundance, (b) species richness and (c) abundance/species richness index for samples falling inside (i.e., treatment T) and outside (i.e., reference R) the licensed aggregate extraction area. Annual means with 95% bootstrapped confidence intervals for pairwise comparisons of ratios of (d) abundance, (e) species richness and (f) abundance/species richness index calculated between treatment (T) and reference (R) stations. Dotted lines denote the upper and lower limits for acceptable change (as described by Rees and Pearson (1992)), solid lines at zero indicate equality of status.

localised dredging and screening for gravel may therefore be expected to alter the sediment profile of the nearby benthic environment. Evidence for this has become apparent at Cross Sands judging by the significantly lower mean sorting coefficient for sediments inside the licensed extraction area. A lower sorting coefficient is indicative of a more homogeneous mix of particle size classes, such as what would result after removal of the gravel fraction. Principal Component Analysis (Fig. 2) illustrates how stations falling within the licensed extraction areas are differentiated along the axis corresponding to the medium and coarse sand components of the sediment, whilst stations outside the area are differentiated along other axes. It seems unlikely that under the present hydrological conditions – where only the lighter sandy fractions are transported in the currents (Desprez, 2000) – the altered sedimentary profile will return to its pre-dredged heterogeneous state.

A reduction in heterogeneity similar to that observed for the sediment profile was also detectable in the faunal assemblage inside the licensed extraction area. Although faunal samples taken from outside the licensed extraction area could be as diverse/impoverished as those taken from inside the extraction area, the converse situation was not observed (Fig. 4). This observation may seem at odds with the perceived wisdom postulated by Warwick and Clarke (1993) that increased disturbance of natural marine communities leads to an increase in heterogeneity. It is likely, however, that the natural condition of the seabed around Cross Sands is already fairly dynamic and therefore variable. Frequent disturbance of the sediment within the licensed extraction area effectively dampens the existing naturally high variability in assemblage structure by removing and/or preventing the re-establishment of a more mature and diverse benthic assemblage. As a consequence, variability appears reduced.

In terms of faunal abundance, species richness and univariate measures of diversity, impacted benthic assemblages may take at least a decade to recover to pre-dredged levels (Boyd et al., 2005), especially since an important structural and stabilising element of the sedimentary habitat has been removed. However, this does not preclude functional recovery of the benthic assemblage over shorter time scales. Research is presently underway to better understand the effects of aggregate extraction on benthic ecosystem function and to monitor the dynamics of the recovery process.

4.2. Temporal variability and recommendations for environmental monitoring

Despite being relatively less variable and diverse, faunal assemblages inhabiting dredged sediments appear to be more dynamic than their undisturbed counterparts (as evidenced by the significant differences in certain assemblage parameters between years – Table 2). Assemblages inside the licensed extraction area may be in a state of flux, possibly evolving towards a new state of equilibrium after disturbance, or unable to reach maturity owing to the unstable nature of the remnant sediments. Whichever the case, it is not the passage of time that has the biggest influence on the composition of the benthic assemblage but the combination of physical and environmental conditions present at any particular station (as shown by the higher Rho values from BIOENV tests relative to those from RELATE – Table 3). Only at G23 did the passage of time appear to be more influential than the physical properties of the sediment, yet, as has already been noted, G23 was subjected to abnormally intense dredging activity (Table 1). It seems likely that the temporal variation observed in the macrofaunal assemblage at G23 was a direct result of disturbance by frequent and intensive dredging, and not necessarily a reflection of natural temporal fluctuations in population dynamics.

The net effect of dredging on the faunal assemblage from year to year appears to be variable and irregular, as illustrated in Fig. 5d–f, without any suggestion of a developing trend over the 8-year study period. Such behaviour not only highlights the unpredictable effect of dredging disturbance on the benthos, but also confirms the need for monitoring at a frequency no shorter than once a year. Monitoring surveys conducted at even shorter intervals (e.g., seasonally) may be expected to detect a degree of variability equal to or above that observed during yearly sampling, yet it may prove difficult to separate seasonal variability from that due to disturbance. Alternatively, surveys conducted once every two or three years may arrive at very different conclusions on the effects of dredging (this can be tested by selectively ignoring alternate years in Fig. 5a–f; a different conclusion could be reached depending on whether one chooses to ignore odd or even years). In effect, sampling every two or every three years would only mean that the extent of natural temporal variability in the assemblage would take twice to three times as long to record. The value of long-term data sets is therefore paramount if the environmental impacts of human activities are to be discerned from a naturally changing environment, and the successful development and testing of relevant indicators is to be achieved.

One important aspect of future work in the field of monitoring the ecological effects of aggregate extraction is to ascertain whether there are limits beyond which the capacity of impacted habitats to recover is compromised. After so many years of sustained dredging at Cross Sands, it is encouraging to see that even when one of the measured variables departs significantly from an equitable state, the effect does not persist from one year to the next (Fig. 5d–f). In other words, the potential for short-term partial recovery of the assemblage has not been compromised (at least in terms of abundance and species richness). Such capacity for recovery is also evident even when measured parameters have gone beyond the hypothetical acceptable limits (in this case based on Pearson and Rosenberg's (1978) model on benthic responses to organic enrichment). Obviously, the exact values of acceptable limits for disturbance caused by dredging have yet to be developed, as well as any possible modes of intervention to remedy any critical damage caused. The practical and environmental implications of habitat remediation are already being explored, while incorporating elements of ecosystem function into the monitoring and the recovery processes remains a challenge for the future.

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Recovery of the seabed following marine aggregate dredging on the Hastings Shingle Bank off the southeast coast of England

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Abstract

The aim of this study was to investigate the effect of dredging intensity on the physical and biological recovery times of the seabed following marine aggregate dredging. Two areas of seabed, previously subject to, respectively, relatively high and lower levels of dredging intensity, were identified on the Hastings Shingle Bank. Two reference areas were also selected for comparative purposes. All four sites were monitored annually over the period 2001–2004, using a combination of acoustic, video and grab sampling techniques. Since the site was last dredged in 1996, this was intended to provide a sequence of data 5–8 years after cessation of dredging. However, an unexpected resumption of dredging within the high intensity site, during 2002 and 2003, allowed an additional assessment of the immediate effects and aftermath of renewed dredging at the seabed. The early stages of recovery could then be assessed after dredging ceased in 2003. Results from both dredged sites provide a useful insight into the early and latter stages of physical and biological recovery. A comparison of recent and historic dredge track features provided evidence of track erosion. However, tracks were still visible 8 years after the cessation of dredging. Within the high dredging intensity site, recolonisation was relatively rapid after the cessation of dredging in 2003. Rather than indicating a full recovery, we suggest that this initial ‘colonization community’ may enter a transition phase before eventually reaching equilibrium. This hypothesis is supported by results from the low intensity site, where biological recovery was judged to have taken 7 years. Further monitoring is needed in order to test this. An alternative explanation is that the rapid recovery may be explained by the settlement of large numbers of *Sabellaria spinulosa*. As the resumption of dredging within the high intensity site limited our assessment of longer-term recovery it is not yet possible to assume that a 7-year biological recovery period will be applicable to other, more intensively dredged areas at this or more distant locations.

Keywords: Hastings Shingle Bank; recovery; benthos; aggregate; dredging; seabed; environmental impact

1. Introduction

Currently, around 21% of the supply of sand and gravel in England and Wales comes from the marine environment. In 2005 this demand amounted to 21.09 million tonnes of aggregate (Crown Estate records, unpublished). This material, primarily used in construction and coastal defence, is sourced from licenced extraction areas around the coast of the United Kingdom, using either anchor or trailer suction hopper dredging vessels.

As would be expected, ongoing dredging operations lead to reductions in the numbers of species and individuals in the immediate vicinity within licenced areas (e.g. Shelton and Rolfe, 1972; Kenny et al., 1998; van Dalfsen et al., 2000; Sardá et al., 2000; van Dalfsen and Essink, 2001; Boyd et al., 2003, 2005).

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In addition, physical changes such as the creation of dredge furrows or pits, depending on the method of dredging employed, and alterations to the composition of seabed sediments may also result (Dickson and Lee, 1972; Millner et al., 1977; Kenny et al., 1998; Desprez, 2000; Limpenny et al., 2002; Boyd et al., 2003, 2004). Whilst the severity and persistence of such effects depend on local environmental conditions (e.g. hydrography, geology, and type of benthic community), it is generally assumed that they will disappear, typically on timescales of 1–10 years. In order to promote recovery, licence conditions normally dictate that the seabed be left in a 'similar' condition to that which existed prior to the onset of dredging. As the acceptability of dredging activities is likely to depend on the persistence of any impacts after the event of cessation, as well as their severity and spatial extent while extraction is taking place, it is important that those responsible for the management of such activities have information on recovery-times.

A number of authors have investigated the capacity for the physical and biological recovery of the seabed at Hastings Shingle Bank following dredging (Dickson and Lee, 1972; Shelton and Rolfe, 1972; Rees, 1987; Kenny, 1998). Whilst these studies have made predictions about likely physical and biological recovery-times, none have provided definitive times. As a result, recovery time predictions are often based on available studies from elsewhere. For example, Kenny et al. (1998), working off North Norfolk, suggest that this process could be expected to take around 2–3 years. However, their study followed recovery after a 'one-off' experimental dredging event, rather than the more sustained dredging more commonly associated with commercial sites. In recent years, the availability of empirical data on recovery-times at commercial sites has increased (e.g. Boyd et al., 2003, 2004, 2005) and, in general, suggests that recovery-times may typically be longer than 2–3 years. The variability in the findings from different locations, and in response to different dredging regimes (Boyd et al., 2004), indicates that there is still a need to increase the number of such case studies in order to build a more complete picture of recovery rates around the coast of the UK. The present study was designed to test earlier predictions regarding recovery-times at the Hastings Shingle Bank (Dickson and Lee, 1972; Rees, 1987; Kenny, 1998) with special reference to the effects of dredging intensity.

2. Materials and methods

2.1. Study site

The study site is located approximately 6 nautical miles south of Hastings off the south coast of England (Fig. 1). Water depths vary from 14 to 40 m below chart datum, and the tidal ellipse is aligned in an NE–SW direction. The maximum spring tidal current velocity is 2.6 knots (Admiralty Chart 536). On the flood tide the flow is in a north-east direction, whilst water flows south-west on the ebb. Current meter studies in the area (Rees et al., 2000) and observations of seabed transport features from this study indicate that the net sediment transport is in a north-easterly direction.

Hastings Shingle Bank was first licenced for aggregate dredging in September 1988. Since this time there have been a number of changes to the boundaries of the extraction licence. In 2001, sub-areas X and Y (see Fig. 1) were both relinquished and replaced by a new licence. Whilst this new licence encompasses part of the old sub-area X, some previously dredged areas fall outside, making them suitable for an investigation of recovery. In addition, it was considered unlikely that the northern extreme of the new licence (the area which overlaps with Area X) would be dredged during the period of this study, and so this area was also considered suitable for investigation. However, in 2002 dredging resumed within this area and continued into 2003. Therefore results from the high intensity site in these two years represent conditions within a current aggregate extraction area.

2.2. Sampling design

Since 1993, every vessel dredging on a Crown Estate licence in the UK has been fitted with an Electronic Monitoring System (EMS). This consists of a computer linked to a Global Positioning System (GPS) and one or more dredging status indicators. Every 30 s the computer automatically records the date, time and position of all dredging activity. EMS information was interrogated in order to locate areas of the seabed within Area X which had been subjected to different levels of dredging intensity.

Areas of high and lower levels of dredging intensity were identified from the 1996 EMS data (see Fig. 2), the last year that the study site had been dredged prior to sampling in 2001. The area of high dredging intensity represented >4.99 h of dredging whereas the area of lower dredging intensity is equivalent to <1 h of dredging, within each 100 m by 100 m block. Treatment boxes, measuring 300 m by 200 m, were assigned to these two areas of seabed (see Fig. 1).

Two reference sites were also selected, using side-scan sonar and video images of the seabed. Note that these sites were not considered representative of 'baseline' conditions, as there was insufficient information on which to determine what actually constitutes the likely pre-dredging status of the area. However, they were considered to be representative of the wider environment surrounding the extraction licence and outside of the influence of potential dredging effects. The area of each reference box was half that of the treatment boxes.

Within each treatment box, 10 randomly positioned sampling stations were identified (stratified random design). In order to achieve the same sampling density, only five random stations were identified within each reference site. All station positions were re-randomised each year and details of the locations sampled are presented in Table 1.

2.3. Sample collection

One 0.1 m^2 Hamon grab sample was collected from each randomly positioned sampling station within the high and

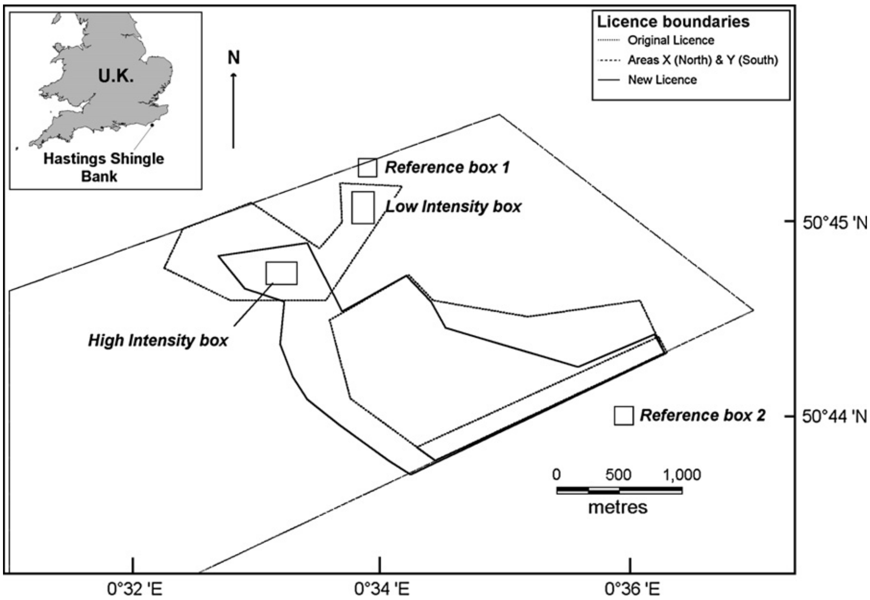

Fig. 1. Treatment boxes in relation to positions of historic (original, Area X and Area Y) and current (new licence area) aggregate extraction licence on the Hastings Shingle Bank.

low intensity sites and the reference sites in each year of the study (2001–2004). Samples were collected in July aboard the RV Cirolana (2001 and 2002) and RV Cefas Endeavour (2003 and 2004). Following estimation of sample volume, a 500 ml sub-sample was removed for laboratory particle size analysis. The residual sediment was then washed over a 1 mm square mesh sieve to extract the macrofauna, which was then back-washed into a watertight container and fixed in 4–6% buffered formaldehyde solution (diluted in seawater).

2.4. Acoustic and video surveys

A side-scan sonar survey was undertaken using a Datasonics™ SIS 1500 digital chirp system in July of each year

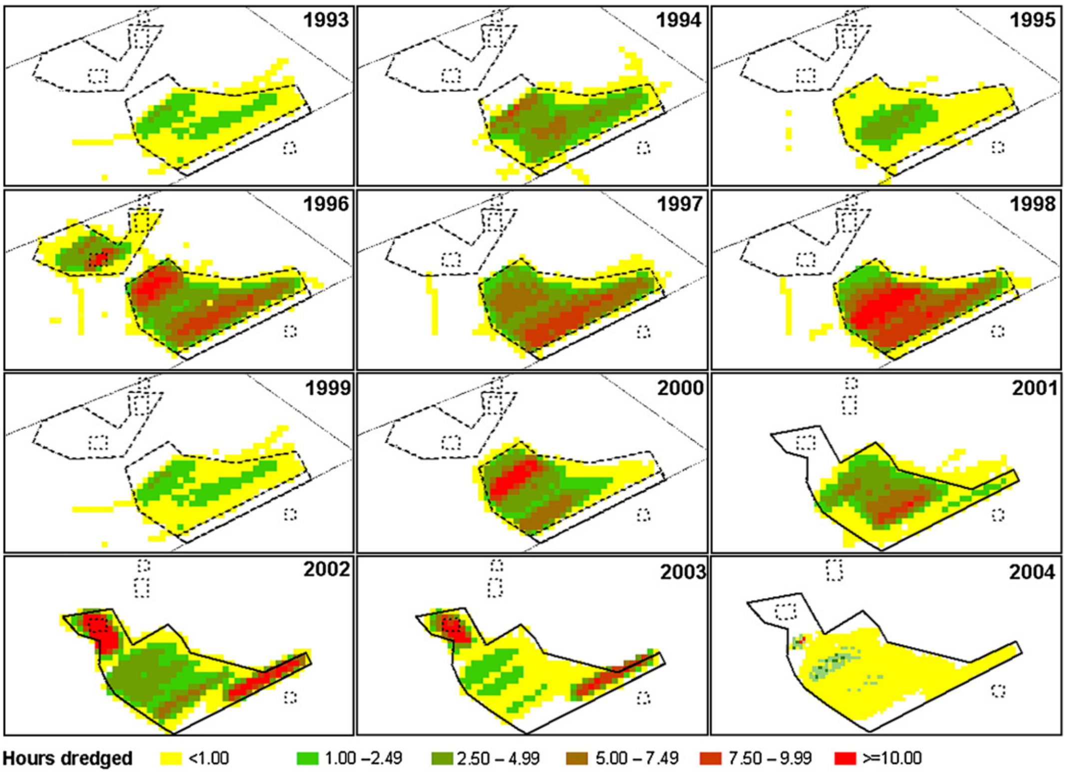

Fig. 2. Location and intensity of dredging (hours) over each 100 m × 100 m block at Hastings Area X in relation to the positions of the high and low sampling sites in years 1993–2003 (source Electronic Monitoring System data provided by the Crown Estate).

Table 1
Co-ordinates of treatment boxes

Treatment	Code	Box co-ordinates		Area (m^2)	Number of samples collected			
		Latitude	Longitude		2001	2002	2003	2004
High intensity site	HIGH '01–'04	50° 44.784′N 50° 44.670′N	00° 33.090′E 00° 33.336′E	~60,000	10	10	10	10
Low intensity site	LOW '01–'04	50° 45.144′N 50° 44.982′N	00° 33.780′E 00° 33.960′E	~60,000	10	10	10	10
Reference site 1	REF '01–'04	50° 45.314′N 50° 45.221′N	00° 33.833′E 00° 33.980′E	~30,000	5	5	5	5
Reference site 2	REF '01–'04	50° 44.046′N 50° 43.954′N	00° 35.898′E 00° 36.047′E	~30,000	5	5	5	5

(2001–2004). The purpose of these surveys was to identify the spatial distribution of superficial sediment types and bed-forms across the current and relinquished zones of the Hastings Shingle Bank extraction area and also across the wider region. Thirteen survey lines (approximately 5 km long) were surveyed in a north–south orientation using a 400 m line spacing in order to achieve 100% coverage of the survey area. The digital data were acquired and post-processed using the Triton Isis™ and Delphwin™ software packages, producing a geo-referenced, on-screen mosaiced image of the survey lines.

Multi-beam surveys were carried out using a dual-head, hull-mounted, Kongsberg Simrad EM 3000D high-resolution multi-beam sonar. The purpose of these surveys was to provide detailed bathymetric data in order to monitor the dimensions of dredge tracks over the period. The data were corrected in real time for vessel movements using a Kongsberg Seatex Motion Reference Unit (MRU 5). Soundings were acquired using TEI Inc, Triton Isis™ software and the data were tidally corrected and gridded using the TEI Inc, Bathypro™ processing package. The data were presented using IVS3D Fledermaus™ software. Given the depths of water encountered, multi-beam swathe widths were typically half of the side-scan sonar swath widths.

Where conditions allowed, photographic surveys using underwater video and stills techniques were conducted using a Simrad™ video camera and a Benthos DSC™ 4000 digital stills camera mounted within a robust metal frame. These surveys were used to obtain additional ground-truth information on the physical and biological status of the seabed. The camera frame was lowered close to the seabed as the vessel drifted with the tide. Video images were recorded automatically onto high-resolution digital tape. Deployments of approximately 10 min duration were carried out over the high and low intensity sites, and also the reference sites.

2.5. Sample processing

2.5.1. Macrofauna

In a fume cupboard, the formaldehyde solution was removed by draining the sample over a 1 mm mesh sieve. The sample was then subjected to a series of washes with fresh water, which served to remove any remaining formaldehyde and also to remove any lighter animals. Small aliquots of the remaining sediment were then transferred to a white plastic tray and examined under an illuminated magnifier in order to remove any remaining animals. Specimens were placed into a labeled Petri-dish, containing a preservative of 70% Industrial Methylated Spirits. The animals were then identified to the lowest possible level, usually species, and enumerated. Finally, the blotted wet weight (in milligrams) for each species, from replicate samples, was recorded.

2.5.2. Sediment particle size

The sediment sub-samples from each grab were analysed for their particle size distributions. Samples were first wet-sieved on a 500 μm stainless steel test sieve using a sieve shaker. The <500 μm sediment fraction passing through the sieve was allowed to settle from suspension in a container for 48 h. The supernatant was then removed using a vacuum pump and the remaining <500 μm sediment fraction was washed into a Petri-dish, frozen for 12 h and freeze-dried. The total weight of the freeze-dried fraction was recorded. A sub-sample of the <500 μm fraction was then analysed using a laser sizer. The >500 μm fraction was washed from the test sieve into a foil tray and oven dried at ~90 °C for 24 h. It was then dry-sieved on a range of stainless steel test sieves, corresponding to 0.5 phi intervals, down to 1 phi (500 μm). The sediment on each sieve was weighed to 0.01 g and the values were recorded. The results from these analyses were combined to give a full particle size distribution for each sample.

2.6. Data analysis

Analyses techniques were chosen to determine:

(1) whether there were statistically significant differences between the dredged and reference locations, and, if so;
(2) whether there was any evidence of a trend towards increasing similarity over time.

2.6.1. Sediment variables

Particle size data are summarized as annual means taken from the high and low intensity sites and also the reference sites. In addition, multivariate techniques (Principle Component Analysis and ANOSIM) were used in order to identify differences between the sediment particle size composition of samples from high and low dredging intensity and reference sites.

2.6.2. Macrofaunal assemblage structure

Ash Free Dry Weights (AFDW) were calculated using standard conversion factors (Ricciardi and Bourget, 1998). The univariate measures of total abundance (N), numbers of macrofaunal species (S) and biomass (AFDW) were calculated and plotted over time. This allowed a visualisation of any trends (e.g. increasing or decreasing abundance at different sampling locations and over time). The significance of differences between sites was tested using one-way ANOVA.

All multivariate analyses were performed using the software package PRIMER v. 6.1.5 (Clarke and Gorley, 2001).

3. Results

3.1. Sediment characteristics

Differences, in terms of mean particle size composition, between the high and lower dredging intensity and the reference sites in each year of the study are shown in Fig. 3a and within each site over the course of the investigation in Fig. 3b. In 2001, all sites showed a high degree of similarity, especially between the site of lower dredging intensity and reference sites. This observation was confirmed by the results of an ANOSIM test. However, a number of predominately sandy samples, not encountered at the reference sites, were found within both dredged areas. During 2002 and 2003 dredging resumed within the high intensity site and sediments became coarser in comparison to the lower intensity and reference sites, which remained similar to one another. By 2004 (after the cessation of dredging) the high dredging intensity site had become sandier and in general, more similar to the lower intensity and reference sites.

3.2. Acoustic and video surveys

3.2.1. Broad scale spatial survey

Fig. 4 shows a mosaic of side-scan sonar data from 2002 upon which is superimposed the boundaries of the current and historic licenced extraction areas. Also shown are the high and low intensity and reference sites (see Fig. 1). Within the current licence, gravely sediments are intensively furrowed by tracks formed by suction hopper trailer dredgers. Surrounding the licenced zones are a number of relatively undisturbed areas of stable sandy gravel, indicated by darker uniform patches on the side-scan mosaic. The two reference sites from the present study are sited within two such areas, to the north and south-west of the current licence. Similar areas are present immediately to the north-east of the extraction licence and in the extreme south-west of the survey area. In contrast, areas of sand, as indicated by the lighter patches on the side-scan mosaic, are found to the north-east and extreme north-west of the licence. In the extreme north-east of the survey area the sandy substratum is formed into large sand waves. Other areas surrounding the current licence are characterised by thin discontinuous veneers of sand overlying coarser deposits. These observations are consistent with other studies in the area (Brown et al., 2004).

3.2.2. Temporal investigation of study sites

Fig. 5(a–d) shows side-scan sonar images collected from the high and lower dredging intensity sites and reference site 1, in 2001 and 2002. In 2001, weathered dredge tracks, orientated inan NE–SW direction, were clearly visible within the high intensity site (Fig. 5a). The features were in-filled with sand and it appeared that they may have formed from the agglomeration of individual tracks. In 2002, recent dredging activity (characterised by a generally N/S track orientation) had begun to mask these historic dredging-related features (Fig. 5b) and in 2003 they had all but disappeared. Video images revealed that the recent tracks were characterised by steep ridges of clean gravel on either side of the track, the bases of which were in-filled with sand (see Fig. 6a). The dimensions of a characteristic track from this area, determined using multi-beam bathymetric data were estimated to be $4 \text{ m} \times 0.24 \text{ m}$.

Weathered dredge tracks are apparent from the side-scan sonar image within the lower intensity site in all four years, but to a far lesser degree than in the high dredging intensity site (Fig. 5c). Video images collected in the lower intensity site in 2003 were similar to those collected in 2001, with a smooth, flat sediment profile comprising of flat sandy gravel occasionally masked by sand veneers (Fig. 6b). In contrast to the recent dredge tracks found in the high intensity site, the historic tracks found within the low intensity site were wider and shallower ($6.5 \text{ m} \times 0.10 \text{ m}$). These tracks are estimated to be at least 7 years old, according to the EMS data.

Side-scan sonar images of the reference sites show that the surrounding sediments are generally similar between 2001 and 2004. The seabed consists of flat generally featureless sandy gravels, with occasional sand veneers (Fig. 6c). In 2002, the side-scan sonar image provides some evidence that demersal fishing activity, indicated by paired tracks, has occurred in this area (Fig. 5d).

3.3. Macrofaunal assemblage structure

Overall, a total of 457 taxa were found at Area X from the 120 samples collected between 2001 and 2004. The numbers encountered annually were 268, 328, 268 and 306.

3.3.1. Univariate analyses

Fig. 7 shows a comparison of various univariate measures at each sampling site in 2001–2004. In 2001, 5 years post cessation, significantly lower ($p > 0.05$) numbers of species were found at the high intensity site in comparison with the reference sites. In addition, although abundance values were not significantly different between these two sites, this could be attributed to elevations of the barnacle *Balanus crenatus* from within the site of high dredging intensity. This had the effect of masking reductions in the abundance of many other species at this site. *B. crenatus* appears to fulfill a role as an opportunistic colonizer of gravel substrata exposed during the dredging process. Following dredging within the high intensity site in 2002 and 2003, the numbers of species, individuals and biomass remained significantly lower in comparison with the

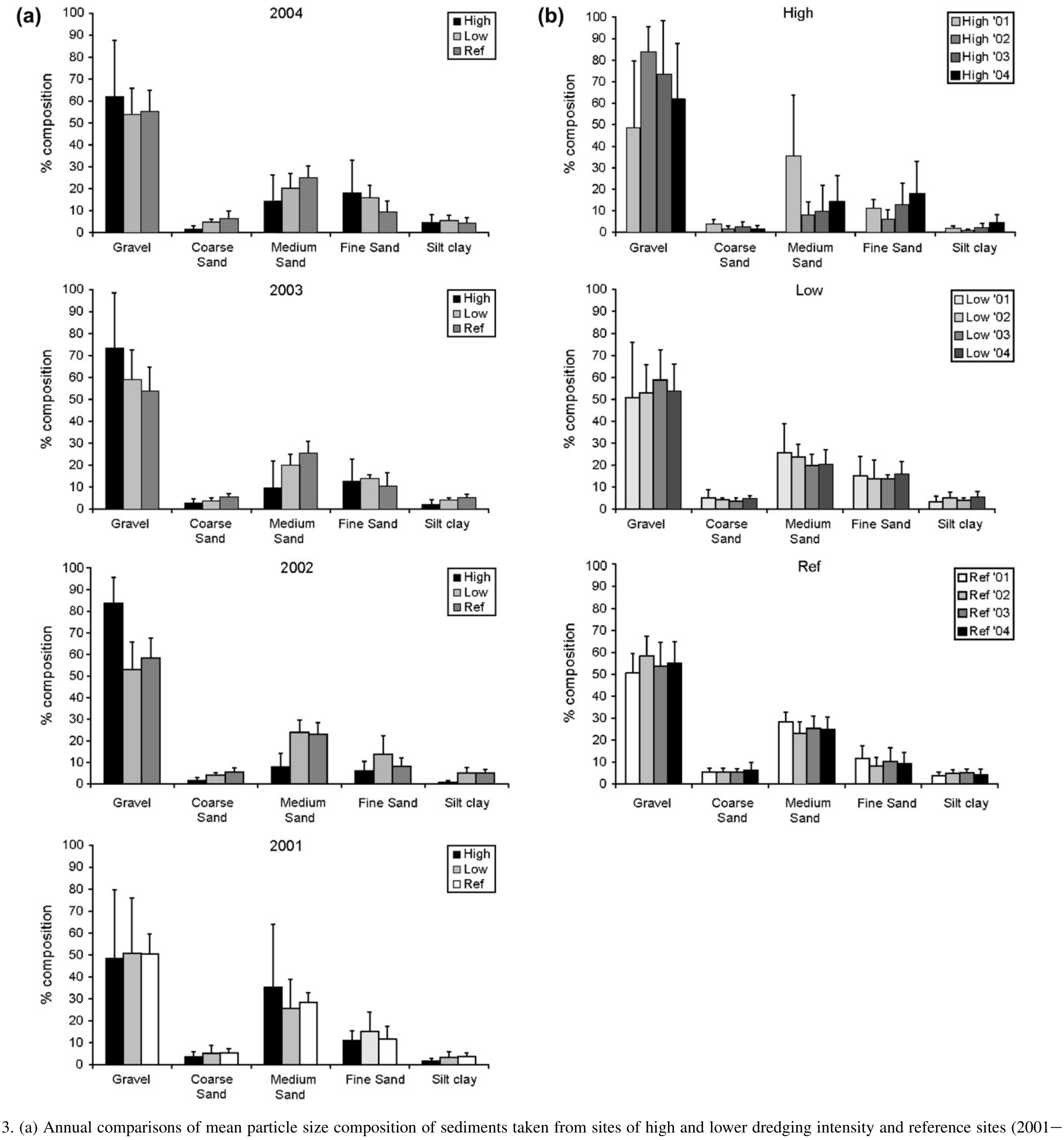

Fig. 3. (a) Annual comparisons of mean particle size composition of sediments taken from sites of high and lower dredging intensity and reference sites (2001–2004). (b) Annual records from each site are displayed together to allow inspection of the between year variation in average sediment composition (2001–2004).

reference sites. However, in 2004, values of all measures rose dramatically such that they did not differ from the reference sites. Of particular interest was the rise in mean abundance, mainly as a result of large numbers of juvenile *Sabellaria spinulosa*, a polychaete worm (Ave. 133.7 ± 278.6) within several samples. This may have been due to a shift in sediment composition to a gravelly/sandy habitat, which is known to be suitable for *S. spinulosa* colonization (Foster-Smith and Hendrick, 2003).

In 2001, numbers of species within the low dredging site were also significantly lower ($p < 0.05$) than the reference sites. In addition, high densities of *Balanus crenatus* were also found within a few of the more gravelly samples obtained from this site. Despite this local enhancement of *Balanus*, the total density of macrofauna was lower from within the area of low dredging intensity compared to elsewhere. However, by 2002, no significant differences ($p < 0.05$) were detectable between the low dredging intensity and reference sites in terms

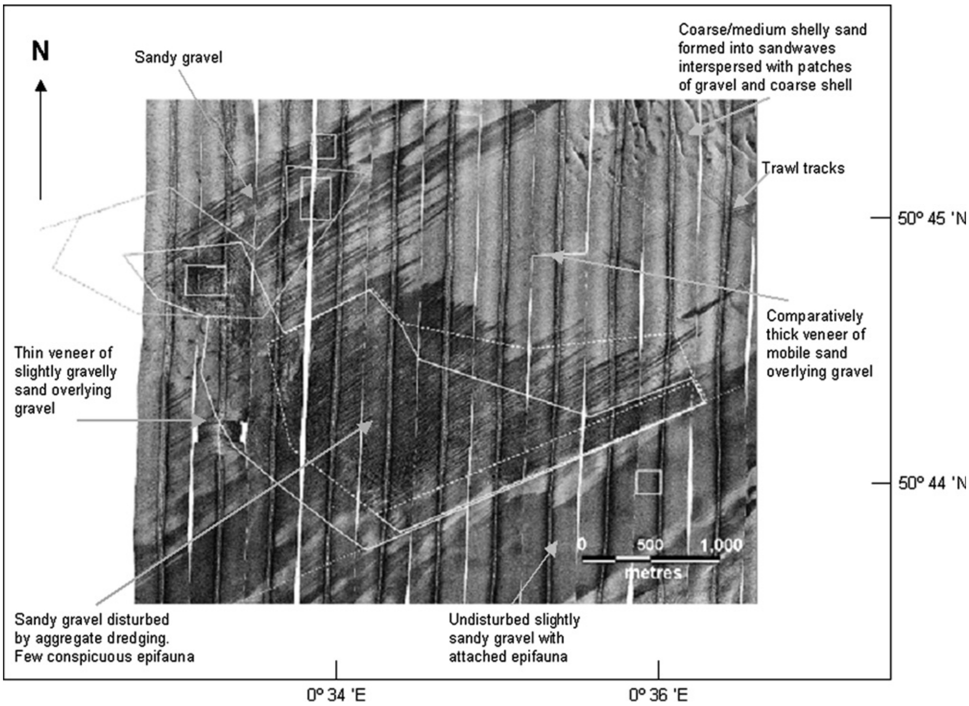

Fig. 4. Side-scan sonar mosaic showing the distribution of substrate types within and surrounding the current and relinquished extraction areas on the Hastings Shingle Bank (see Fig. 1 for licence boundaries).

of number of species and individuals, and in 2003 biomass values were also not significantly different from reference values. This suggests that, in terms of univariate summary measures, the low dredging intensity site had recovered following 7 years after cessation of dredging.

3.3.2. Multivariate analyses

The MDS ordination (Fig. 8) indicates a large degree of overlap of samples from the areas of low dredging intensity and the reference sites. Comparison of ANOSIM *R*-values (Table 2) shows that by 2003 no significant difference

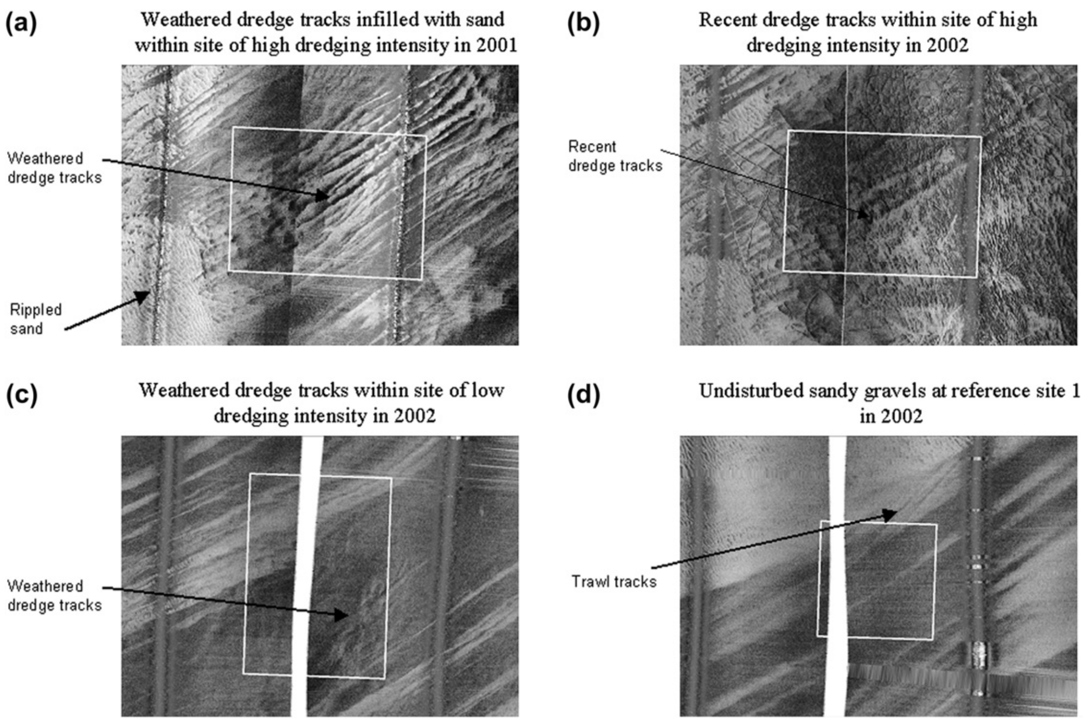

Fig. 5. (a–d) Examples of side-scan sonar records from high and low dredging intensity sites and reference site 1.

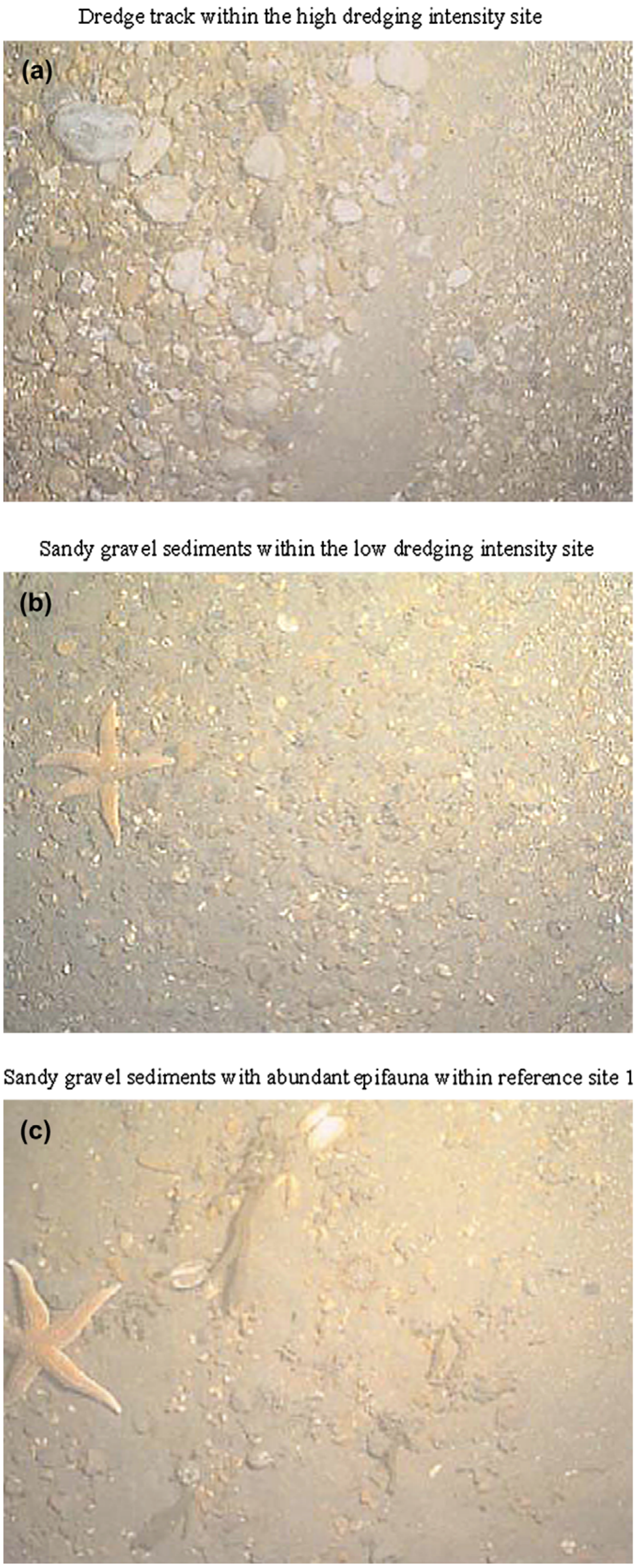

Fig. 6. (a–c) Underwater photographic images taken from sites of high and lower dredging intensity and reference site 1 in 2003. Each image represents an area of seabed of approximately 1.4 m by 1.0 m.

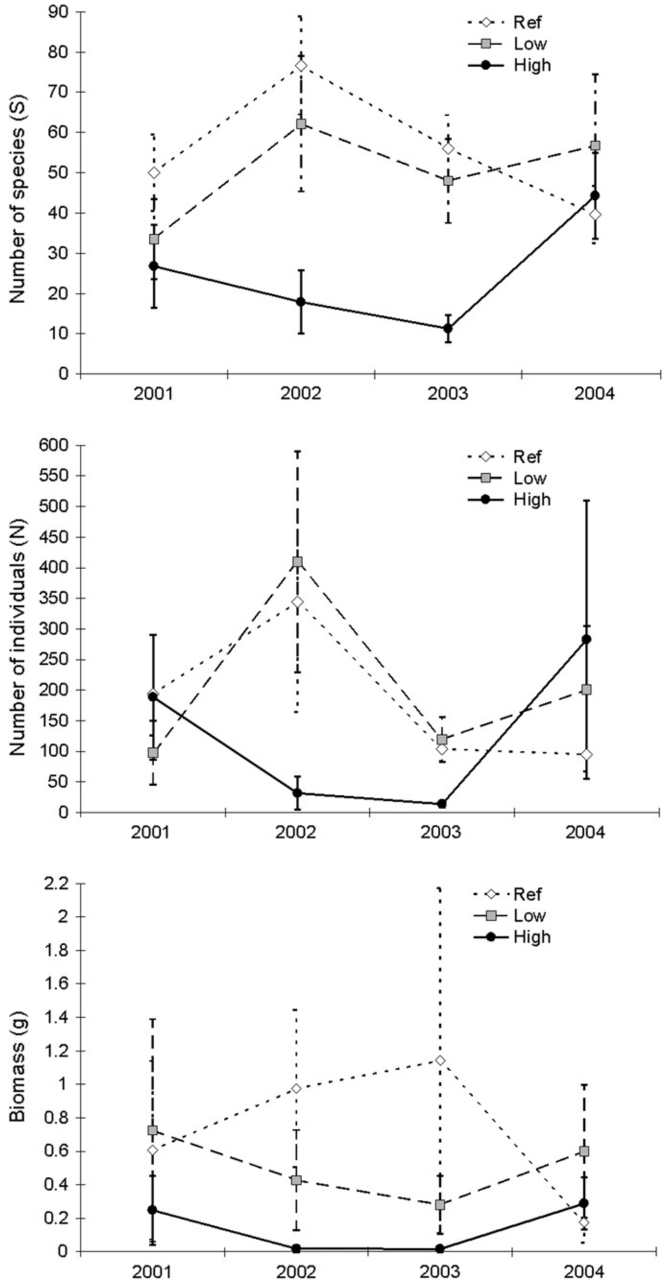

Fig. 7. Summary of means and 95% confidence intervals for numbers of species (S), number of individuals (N) and biomass (AFDW(g)) from sites of high and lower levels of dredging intensity and two nearby reference sites between 2001 and 2004.

($p > 0.05$) could be detected between the low dredging intensity and the reference sites. This suggests that restoration of the fauna was achieved in those parts of Hastings Area X exposed to lower levels of dredging intensity after a period of approximately 7 years since the cessation of dredging.

In 2001, 5 years after cessation of dredging, over half of the samples from the high dredging intensity site were present in the main cluster of the MDS. Samples are more diffusely separated on the MDS in 2002 and 2003, whilst dredging took place. This suggests that they are biologically dissimilar to samples collected elsewhere and reflect the substantially

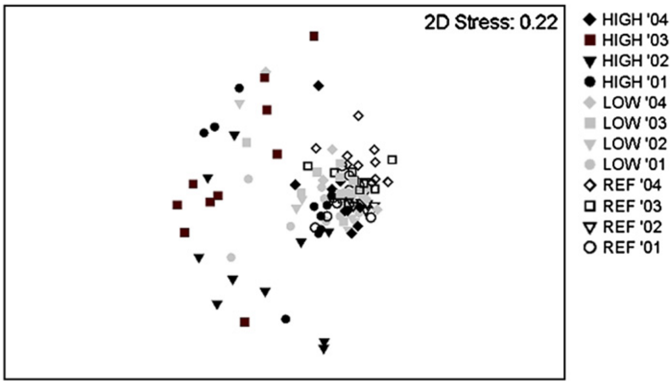

Fig. 8. MDS of Bray–Curtis similarities from fourth root transformed species abundance data (colonial species excluded) at the high and lower dredging intensity sites and at the reference sites (data from 2001–2004).

reduced densities of organisms present. By 2004, less than 12 months after the last dredging episode, only one high intensity site sample is found outside the main cluster. Despite this, ANOSIM values indicate a persistent difference between the high intensity and reference sites. This value, however, may be exaggerated by an apparent shift in the reference sites, as shown on the MDS ordination. For this reason a comparison with the low intensity site may be more meaningful and this indicates only a small difference between high and low intensity sites.

Community groupings were further explored using the similarity percentages Programme SIMPER. Table 3 shows the characterising species of the study sites in 2004. Generally, the fauna at all sites was characteristic of sandy gravel sediments. The lower dredging intensity and reference sites showed similar community structures across all years, although the importance of the characterising species for each site was variable over time. The high dredging intensity site exhibited a decrease in the abundance of the opportunistic colonizing species *Balanus crenatus* during 2002 and 2003 following the resumption of dredging within this area. In 2004, 12 months after dredging had ceased, *B. crenatus* had again become established within the high dredging intensity site along with the tube-dwelling polychaete, *Sabellaria spinulosa*.

4. Discussion

By sampling areas previously subject to high and lower levels of dredging intensity, the survey was designed to allow investigation of the effects of dredging intensity, both in terms of the severity of impact and subsequent recovery-times.

At the outset of this work in 2001 it was anticipated that, following cessation of dredging in 1996, the study would provide information on the status of the seabed, within both dredged sites, 5–8 years after cessation. Whilst this was the case for the low intensity site, dredging resumed, during 2002 and 2003, within the high intensity site. Although not foreseen, this resumption allowed an assessment of the immediate effects of ongoing dredging on the seabed, and its aftermath 1 year after cessation. Together, the results from both sites provide a useful insight into the processes leading to recovery of the seabed following marine aggregate dredging at this site.

4.1. Physical effects and recovery

The effects of ongoing dredging activity were visible at the high intensity site in 2002 and 2003, where the seabed appeared extremely uneven as a result of north-south orientated dredge tracks. This contrasts with the flat seabed observed at the reference sites. Dredge tracks were composed of steep-sided gravel ridges separated by occasional sand-filled troughs. Despite this localised trapping of sand, particle size data revealed a coarsening of sediments during dredging, possibly as a result of the exposing of coarser deposits or the mobilisation of sand away from the site. This phenomenon was also observed by Kenny and Rees (1996) at an experimental dredged site off North Norfolk.

Older dredge tracks, seen in the high intensity site in 2001, and the low intensity site in all years (2001–2004), were different in character and provided evidence of weathering. The shape of these features suggests the furrow sides have collapsed, resulting in a widening and shallowing of the track. Individual tracks also appeared to have coalesced into larger features, and also appeared to trap sand. As a result of this weathering the seabed appeared much less uneven in comparison with the high intensity site during 2002–2004. However, the continued presence of these features, 8 years after cessation of dredging, suggest they may be long-lived, particularly given the classification of this site as one of relatively 'low energy' (Boyd et al., 2004). This assertion is supported by the persistence of dredge tracks at Area 107 off North Norfolk, where tracks were still visible after 7 years (Limpenny et al., 2002), and Area 222 in the Thames estuary where tracks were shown to be >10 years old (Boyd et al., 2004). Based on water depth, tidal current and wave data, Areas 222 and 107 are considered to be sites of 'moderate' energy (Boyd et al., 2005).

Despite the topographical changes observed above, and the occasional sandy sample encountered, particle size of sediments at both dredged sites in 2001, 5 years after cessation, was similar to the reference sites. In addition, the particle size composition of sediments within the high intensity site in 2004 was similar to reference conditions only 1 year after cessation. This accords with government requirements for the seabed to be left in a similar condition post-dredging

Table 2
R-values derived from the ANOSIM test for macrofaunal assemblages from locations of high and lower dredging intensity and from the reference sites at Area X in 2001–2004. * Denotes significant difference at $p < 0.01$; ** denotes significant difference at $p < 0.05$

Year	HIGH/REF	LOW/REF	HIGH/LOW
2001	0.178**	0.120**	0.091*
2002	0.637**	0.158**	0.459**
2003	0.602**	0.056	0.568**
2004	0.572**	0.222**	0.250**

Table 3
Results from SIMPER analysis of macrofaunal data from Area X (all taxa excluding colonial species, fourth root transformed), listing the main characterising species from samples subject to differing levels of dredging impact in 2004. Average abundance, average similarity and the % contribution to the similarity made by each characterising species are shown. Also listed are the cumulative percentage and the overall average similarity between replicate samples from within each group

Group	Taxon	Average abundance	Average similarity	Similarity/SD	% Contribution	Cumulative %	Overall average similarity
HIGH ’04	*Sabellaria spinulosa*	133.70	2.74	1.61	7.03	7.03	**39.03%**
	Balanus crenatus	47.80	2.44	1.15	6.25	13.27	
	Poecilochaetus serpens	3.70	2.14	1.60	5.49	18.76	
	Nemertea	2.20	1.80	1.77	4.62	23.38	
	Galathea intermedia	2.70	1.78	1.79	4.56	27.94	
	Ampharete lindstroemi	2.50	1.51	1.17	3.88	31.82	
	Glycera tridactyla	1.50	1.36	1.15	3.48	35.30	
	Lumbrineris gracilis	3.30	1.33	1.23	3.42	38.72	
	Scalibregma inflatum	3.00	1.28	1.21	3.29	42.01	
LOW ’04	*Lumbrineris gracilis*	9.00	1.89	1.77	4.79	4.79	**39.55%**
	Upogebia (juv.)	9.00	1.74	1.88	4.41	9.20	
	Pomatoceros lamarcki	8.10	1.62	1.82	4.09	13.29	
	Caulleriella alata	3.10	1.50	1.64	3.80	17.09	
	Mysella bidentata	8.90	1.45	1.15	3.67	20.76	
	Echinocyamus pusillus	3.90	1.32	1.04	3.35	24.10	
	Poecilochaetus serpens	9.60	1.24	1.20	3.15	27.25	
	Scalibregma inflatum	6.20	1.23	1.20	3.10	30.35	
	Nemertea	3.70	1.19	1.16	3.02	33.37	
	Mediomastus fragilis	4.70	1.12	1.21	2.83	36.20	
	Goniada maculata	1.50	1.11	1.07	2.81	39.01	
	Pholoe baltica	2.70	1.11	1.19	2.80	41.81	
REF ’04	*Echinocyamus pusillus*	20.40	4.59	4.97	12.65	12.65	**36.28%**
	Praxillella affinis	3.20	2.69	5.69	7.41	20.06	
	Mysella bidentata	6.80	2.60	1.61	7.18	27.24	
	Lumbrineris gracilis	2.40	2.06	1.84	5.67	32.91	
	Notomastus	2.60	1.98	1.85	5.46	38.38	
	Nemterea	2.00	1.75	1.21	4.83	43.20	

thus are maximising the potential for biological recovery. These results contrast with another dredge site, Area 222, where the high dredging intensity site was observed to be sandier, possibly as a result of the screening of dredged cargoes at this site (Boyd et al., 2005). Screening is the practice of returning unwanted sediment fractions to the seabed and is not permitted on the Hastings Shingle Bank.

4.2. Biological effects and recovery

The effects of ongoing dredging on macrofaunal communities, evident from samples taken within the high intensity site in 2002 and 2003, included a reduction in both the numbers and variety of taxa, in agreement with other studies (Shelton and Rolfe, 1972; Kenny et al., 1998; van Dalfsen et al., 2000; Sardá et al., 2000; van Dalfsen and Essink, 2001; Boyd et al., 2003, 2005).

Results from the high dredging intensity site in 2004 provided information concerning the early stages of recovery. Less than 1 year after dredging operations had ceased, samples showed substantial increases in abundance, number of species and total biomass. Particularly interesting were the large numbers of juvenile *Sabellaria spinulosa* found within this site in this year. In addition, samples from both high intensity and reference sites are closely clustered. Whilst ANOSIM values suggest sizeable difference between the high dredging intensity and reference sites in 2004, the difference is perhaps exaggerated by a subtle shift in reference communities. This highlights a need to improve understanding of natural variability associated with macrofaunal communities found in gravel sediments. Therefore, a more useful comparison in this year can be made between high and low dredging intensity areas, which showed a much greater degree of similarity. Whilst the evidence suggests substantive biological recovery within 12 months, despite the clear topographic differences still evident within the site, it is important to consider these results in the context of those from 2001, 5 years after cessation of dredging, as they suggest conditions remained disturbed at this time.

This effect of dredging-induced disturbance was manifested in a number of ways including:

(1) A **reduced number of species** within the site of high dredging intensity during 2001;
(2) The presence of **opportunistic species** within the dredged sites. Large numbers of *Balanus crenatus* were recorded from the high dredging intensity site in 2001. Were it not for the presence of this species, total abundance values would have been significantly lower than either the low dredging intensity or reference sites. The opportunistic

nature of this species at aggregate extraction sites was also reported by Boyd and Rees (2003).

(3) **Increased variability**, in terms of particle size composition and macrofauna, within the high and low dredging intensity sites. As seen in Fig. 8, there were a number of sample outliers evident from both high and low dredging intensity sites. With one exception, these samples were all associated with predominately sandy sediments. No such samples were encountered at the reference sites and acoustic and video data suggest these samples may be associated with sand trapped within dredge tracks. Increased variability is noted as a feature of disturbance by Clarke and Warwick (1994) and has been reported by Kenny and Rees (1994), Sardá et al. (2000) and Boyd et al. (2005) in relation to aggregate extraction sites.

By 2003, multivariate analyses showed no difference between the low dredging intensity and reference sites. In common with Area 222 in the Thames estuary (Boyd et al., 2005), this suggests recovery of the macrofauna after 7 years. The similar time for recovery observed at these two sites may be linked to a number of similarities between the locations. Firstly, both sites were last dredged in 1996 and were therefore in synchronicity in terms of the recovery period. Secondly, both sites were subject to similar low levels of dredging intensity and thirdly, sediment particle size composition was similar to local reference conditions. In contrast, sediments within the site of higher dredging intensity at Area 222 were finer than those of the low intensity and reference sites (Boyd et al., 2004) and, as a result, recovery appears to be ongoing. These results contrast with a number of other case studies which together suggest that substantial progress towards recovery of the fauna could be expected within 2–4 years following cessation of marine sand and gravel extraction (Kenny et al., 1998; van Dalfsen et al., 2000; Sardá et al., 2000; ICES, 2001). However, these studies were based on experimental 'one-off' dredging events and not the sustained dredging more commonly associated with commercial extraction sites.

The apparent contradiction between the relatively rapid (<12 months) progress towards recovery seen in the high intensity site in 2004 and the 7 year period required for recovery inferred at the low intensity site in 2003 requires some further consideration. Models of succession in the marine environment following cessation of environmental disturbance (Newell et al., 1998; Boyd et al., 2005) offer an explanation which fits with the results of the current study. These models show a peak in various indices (e.g. species richness, abundance and biomass) in the early stages following a disturbance. Following this initial peak, values fall and oscillate until finally stabilising with the establishment of an equilibrium community (Newell et al., 1998). Further monitoring is required in order to validate this hypothesis at the Hastings Shingle Bank. Alternatively, rapid seabed recovery may have occurred as a result of rapid colonization by *Sabellaria spinulosa*. Side-scan sonar data together with some limited grab sampling within the high intensity site in 2005 (L.J. Seiderer, Marine Ecological Surveys, pers. comm.) suggest that the juveniles identified during this study have survived to form reef. Foster-Smith and White (2001), in a study of the Wash, noted that the most well developed *S. spinulosa* reefs seen in the area were associated with ground clearly scarred by dredging activity. They suggest this may be a result of the exposing of sediments more suitable for *S. spinulosa* colonization.

Despite the apparent differences between high and low intensity sites in 2001, the resumption of dredging within the high intensity site in 2002 limited our capability to assess the effects of dredging intensity after this time. We therefore recommend caution in assuming that the 7-year recovery figure could be applicable to other more intensively dredged areas of the Hastings Shingle Bank.

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Paper #11

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Cumulative impacts of aggregate extraction on seabed macro-invertebrate communities in an area off the east coast of the United Kingdom

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Abstract

This study investigates whether there is any evidence of a large-scale cumulative impact on benthic macro-invertebrate communities as a result of the multiple sites of aggregate extraction located off Great Yarmouth in the southern North Sea. Forty 0.1 m^2 Hamon grab samples were collected from across the region, both within and beyond the extraction area, and analysed for macrofauna and sediment particle size distribution in order to produce a regional description of the status of the seabed environment. In addition, the data were analysed in relation to the area of seabed impacted by dredging over the period 1993–1998. Areas subject to 'direct' impacts were determined through reference to annual electronic records of dredging activity and this information was then used to model the likely extent of areas potentially subject to 'indirect' ecological and geophysical impact. Results showed the study area to be characterised by sands in the northern half of the survey area, and sandy gravels in the south. The low diversity communities found across much of the survey area were typical of mobile sandy sediments. However, stations located in the southern half and northern extreme of the survey area tended to support higher numbers of species and individuals. This may be due to marginally enhanced stability arising from the higher proportion of gravel found in samples to the south of the extraction licenses and to the presence of *Sabellaria spinulosa* reef in the north. Analysis of data in relation to areas of predicted dredging impact revealed proportionally less gravel and more sand within the 'direct' impact zone, compared to the 'indirect' impact zone. Whilst multivariate analyses of macrofaunal data did not clearly discriminate between dredging impact zones, a comparison of univariate measures revealed significantly lower numbers of species and individuals in areas which have been subject to 'direct' dredging impacts in comparison with 'reference' areas. This provides good evidence of the near-field consequences of dredging. Values of these measures in the 'indirect' zone were intermediate, although not significantly different from the 'reference' zone. We conclude that, although the dominant influence on assemblages in the region is that of sediment instability induced by tidal currents, we cannot dismiss the possibility of a subsidiary influence of dredging activity in the near vicinity of the licensed block and further investigation is warranted.

Keywords: Aggregate dredging; Cumulative impacts; Macrofauna; North Sea

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1. Introduction

In the United Kingdom, marine sediments are an increasingly important source of aggregate for construction and coastal defence (Singleton, 2001). However, the localised distribution of suitable marine gravel deposits has led to intense extraction activity in specific localities, particularly in an area to the east of the Isle of Wight in the central English Channel and also in an area off Great Yarmouth known as the Cross Sands, in the southern North Sea.

The number of licences in these areas has, in recent years, led to concerns regarding the potential for cumulative short and long-term environmental impacts, particularly as extraction activities often overlap with those of other stakeholders such as the fishing industry.

In the UK, environmental impact assessments required under the current 'Government View Procedure' for the licensing of extraction areas since 1989 (ODPM, 2002) have sought to describe and quantify the benthic communities in the vicinity, prior to any dredging taking place. In cases where licences for commercial dredging are issued, periodic surveys of individual sites have allowed the effects of dredging to be investigated over time (Boyd, 2002). This is consistent with international guidance (ICES, 2003).

The direct impacts of dredging on the macrofauna arise from the removal of sediment which, unsurprisingly, leads to reductions in the numbers of species, individuals and diversity of communities (Kenny et al., 1998; Newell et al., 1998, 2004; Desprez, 2000; Sardá et al., 2000; Van Dalfsen et al., 2000; Van Dalfsen and Essink, 2001; Boyd et al., 2003, 2005). In addition, during extraction, material may be discharged from the dredger via overspill chutes or through screening, so that unwanted sediment fractions are returned to the seabed. Whilst relatively little is known about how this material may impact the benthos (Birklund and Wijsman, 2005), Boyd and Rees (2003) identified effects that extended away from the dredged area, suggesting that it may exert a wider influence as a result of dispersion of the resulting 'plume' of material prior to settling on the sea bed. Other studies, including Newell et al. (2004) and Poiner and Kennedy (1984), have reported an enhanced abundance and biomass of macrobenthic species on the periphery of dredging operations, a phenomenon which may be attributable to organic enrichment derived from fragmented marine benthos discharged from with the outwash water (Newell et al., 1999).

To date, most studies at aggregate extraction sites have been confined to the assessment of the impacts associated with separate licensed areas. The potential cumulative effects that arise from combinations of licensed areas in close proximity have received little attention. In order to address this deficiency, the present study involved the conduct of a broad-scale spatial survey designed to characterise the sediments and benthos from across the region, extending well beyond the area of immediate dredging (Fig. 1) impact in order to determine whether any wider cumulative consequences were detectable and, if so, to determine the likely causes. Thus the issue is critically one of 'scale' which may have important consequences for recovery dynamics.

2. Methods

2.1. Study site

The study area was located off Great Yarmouth and Lowestoft on the east coast of the United Kingdom in water depths of between 12 and 46 m Lowest Astronomical Tide (Fig. 2). The tidal ellipse is rectilinear and aligned in a north–south direction with maximum spring tidal velocities reaching 1.4 m s^{-1}. Residual tidal flow is northward across most of the survey area, although the flow moves eastwards approximately 10 miles north of Great Yarmouth (Holt and James, 1999).

The area was first licensed for marine sand and gravel extraction in 1969. Since then, the annual quantity of material extracted increased from around 2.9 million tonnes in 1975 to 8.7 million tonnes in 1989. From 1989 to 1998 annual extraction levels stabilised at around 8 million tonnes, peaking in 1996 at 10 million tonnes. Overall, between 1975 and 1989 a total of 148 255 194 tonnes of material was extracted. The Cross Sands location remains the largest accumulation of marine aggregate extraction licences in the UK, including 15 separate licences covering, at its maximum extent, an area of 340 km^2. The licensed extraction areas off Southwold (Figs. 2,4,5,7) are not included in these extraction statistics.

2.2. Survey design and sampling

The broad scale of this investigation was designed to encompass the totality of potential impacts from dredging at multiple extraction licences and hence went beyond the spatial scales typically investigated at licensed areas. The survey design was based on historical dredging patterns and the likely extent of any sediment plume movement as a by-product of the extraction process.

In June 1998, 40 samples were taken using a 0.1 m^2 Hamon grab deployed from RV 'Cirolana' (Fig. 2). This device was chosen as a result of its success in sampling sediments containing gravel (Oele, 1978). Stations were located using a differential Global Positioning System and the ship's SEXTANT® software that logs the position of sampling. Single samples were collected from each station. The total volume of the grab samples was estimated and a 500 ml subsample of sediment removed for later particle size analysis. The remaining sample was then washed over 5 mm and 1 mm mesh sieves to aid the sorting process. Retained macrofauna were fixed in a 4% formaldehyde solution for later identification and enumeration in the laboratory.

2.3. Extent of dredging

2.3.1. Direct impact

Since 1993 all marine aggregate dredgers working on UK licences have been fitted with Electronic Monitoring Systems (EMS) that automatically record a position

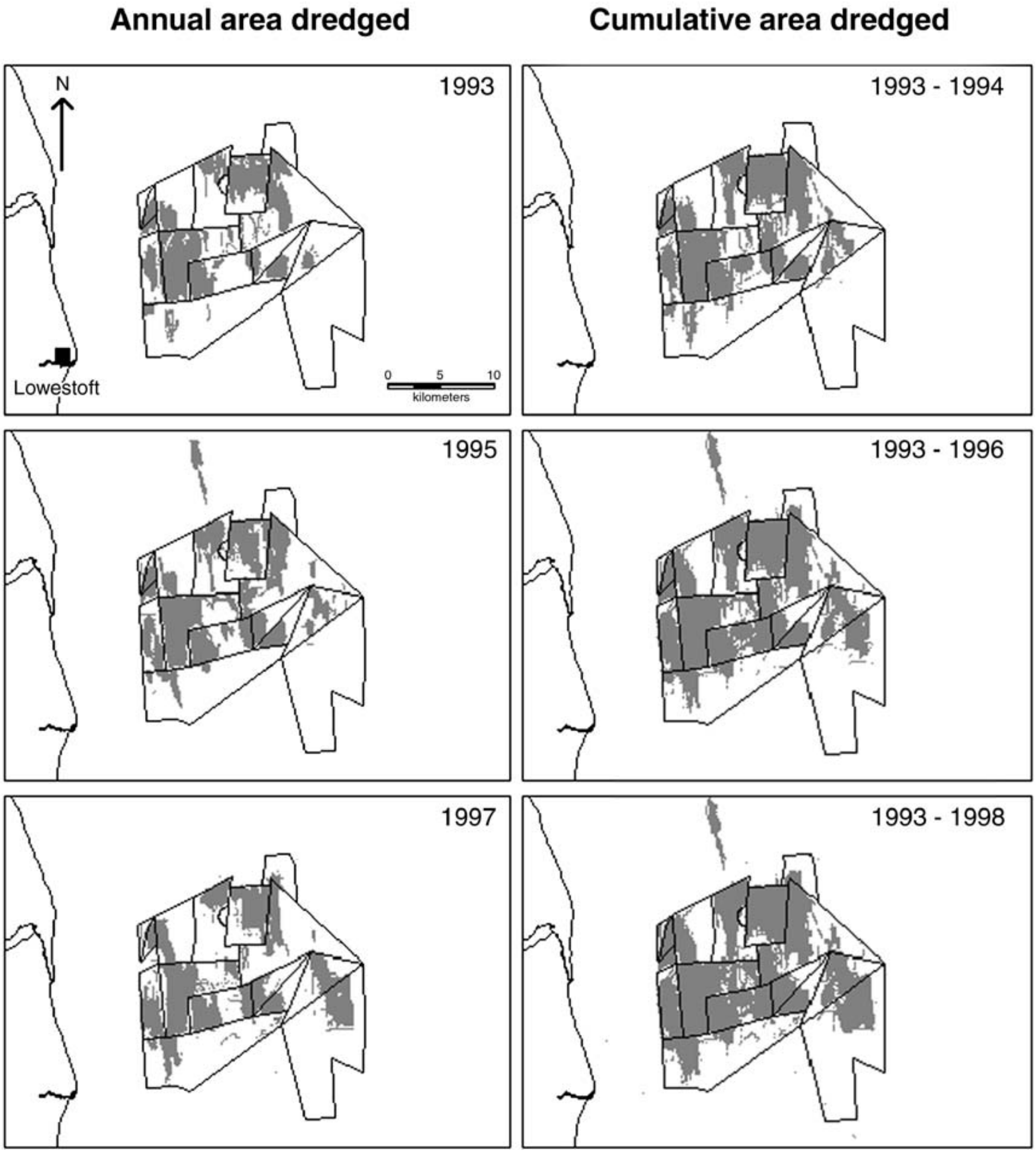


Fig. 1. Location of dredging in individual years and cumulatively since 1993 (based on annual EMS records).

and ‘dredging status’ every 5 seconds to disc. These data are then analysed by the Crown Estate, managers of the UK marine aggregate resource, using a ‘block analysis’ technique to produce maps showing the location and intensity of dredging. These maps show the seabed divided into 100 m × 100 m cells that are coloured according to intensity of dredging. A degree of caution must be exercised in their interpretation, as these blocks are almost certain to overestimate the area dredged. This results from the fact that a 100 m × 100 m block will register as dredged even where it has only been ‘touched’ once by the 2 m wide draghead.

Block analysis maps were obtained for each year from 1993 to 1998 in the form of raster images, which were then input to the Geographic Information System software package Mapinfo®. These images were then geo-referenced and a boundary drawn around the areas impacted by dredging. No attempt was made to differentiate between different levels of intensity, merely the spatial extent of the area dredged. This process was repeated for each year and the records combined to show the cumulative area dredged from 1993 to 1998.

2.3.2. Indirect impact

Whilst good information exists on the location of ‘direct’ impact, areas of the seabed can also be subjected to secondary impacts associated with the discharge of unwanted seabed material back into the water column in a process known as screening. Direct measurements of the spatial extent of screening plumes have been limited to the vicinity of the dredger (Whiteside et al., 1995; Hitchcock and Drucker, 1996; Newell et al., 1998). These studies suggest that a major fraction of the plume material can be expected to be deposited within 300–500 m downstream from the site of dredging. More precise information for the analysis of the data from the present survey was obtained using a Lagrangian particle-tracking model to estimate the dispersal of screened material arising from dredging at licensed areas. The model (EUROSPILL) was originally designed for simulating oil spills (Elliott, 1991) but has been modified subsequently to deal with more general transport problems (Defra, 2001; Perianz and Elliott, 2002).

For the Cross Sands dredging area, the cumulative area dredged was used to define a grid of start locations

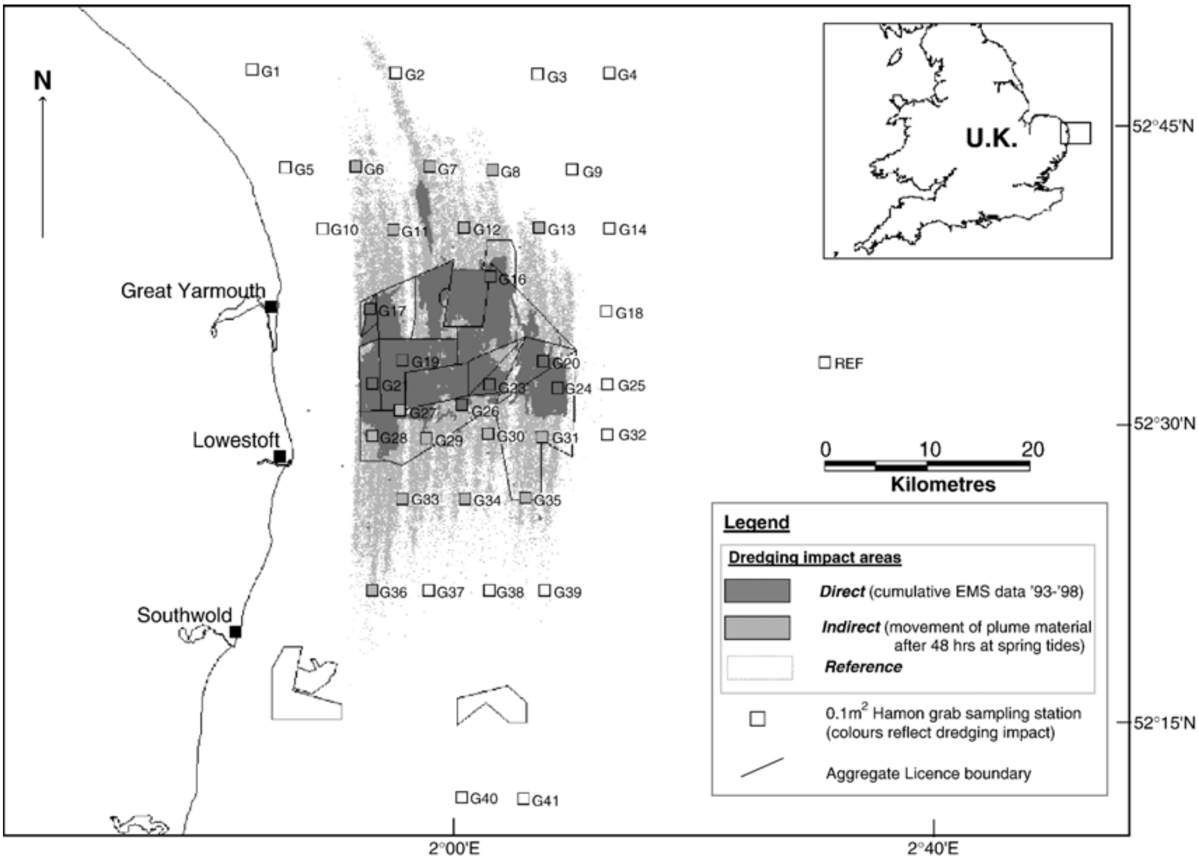


Fig. 2. Location of sampling stations in relation to the distribution of aggregate extraction licences, areas subject to ‘direct’ impact between 1993 and 1998 (based on annual EMS records), areas of ‘indirect impact’ (determined from model output showing all areas which may have been exposed to indirect ‘plume effects’ and reference areas. Licence areas off Southwold are also shown in this figure, but no data are provided on dredging impacts associated with these areas.

covering the licensed area at 0.5 km intervals. At each start position, the movement of 1000 tracer particles, released continuously over a 12.4 hour tidal cycle, were simulated in the model. Separate calculations were undertaken at spring and neap tides to ensure a representative sampling of tidal states. The composition of the dredger overspill plumes is generally unknown, therefore a broad range of particle sizes was considered and simulations assumed a uniform particle size distribution in the range from 20 microns (fine silt) to 200 microns (medium sand). Particles were tracked for a maximum of 48 h and moved horizontally with the local tidal velocity and vertically under the influence of turbulence and a particle-size dependent fall velocity. The position at which each particle hit the bottom was recorded to represent the location of the 'indirect' dredging impact. Clearly such settled plume material could be re-suspended and moved to other locations on subsequent tides. However, because of the difficulty of defining the end of such a process, the dilution in quantity of the material being moved each time, and the increasing uncertainty in predictions, no attempt was made to follow this longer-term fate. Thus, the footprint of 'indirect' dredging impact for the total licensed area was taken to be the totality of the initial settling locations from all start positions.

2.3.3. Allocation of station groups for hypothesis testing

In addition to the 'direct' and 'indirect' zones, Fig. 2 also shows the relative positions of sampling stations from the broad-scale survey. This information was used to assign these stations to one of three groups: 'direct', 'indirect' or 'reference', depending on the likelihood of impacts associated with dredging activity. Thus in our statistical analyses, we tested for differences between

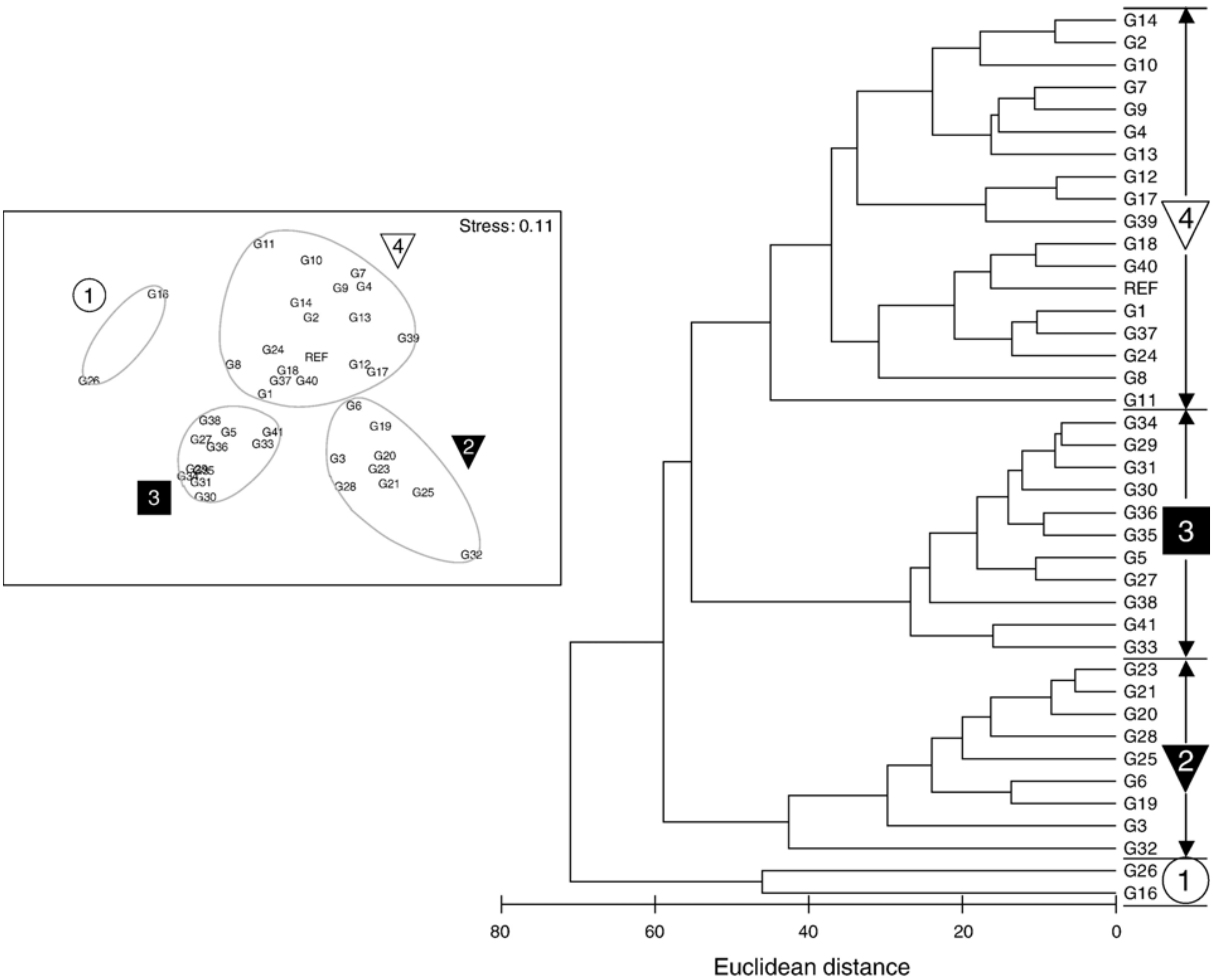


Fig. 3. Dendrogram showing clustering of samples based on percentage particle size composition.

Table 1
Characteristic percentage particle size composition of the particle size clusters identified in the broad-scale survey

Cluster	1	2	3	4
%Gravel	13	10	**50**	9
%Coase sand	11	**55**	10	13
%Medium sand	0	25	20	**48**
%Fine sand	28	8	14	25
%Mud/silt clay	**48**	2	6	5
Average similarity (%)	54.85	76.63	81.31	70.07

'direct' (n=8), 'indirect' (n=15) and 'reference areas' (n=17) where the 'direct' area is expected to show the strongest effects of dredging, 'indirect' effects could be positive or negative and ' reference' areas are considered beyond the influence of the dredging activity.

2.4. Laboratory sample processing

All biological material was stored in a 70% solution of industrial methylated spirits. Collected specimens were then identified to the lowest taxonomic level possible (usually species) and counted. Colonial species were recorded as present.

Sub-samples of sediment from each grab sample were analysed for their particle size distributions. Samples were initially wet sieved on a 500-micron stainless steel test sieve, using a sieve shaker. The <500-micron fraction was then freeze–dried, weighed and a sub-sample analysed using a Coulter LS 130 Laser-Sizer. The >500-micron fraction was oven dried at 80 °C for 12 h and then sieved over a range of test sieves down to 500 microns at 0.5 phi intervals. The sediment retained on each sieve was weighed to the nearest 0.01 g and the results recorded. The results from these analyses were combined to give a full particle size distribution. The mean particle size and the sorting values were also calculated. Sediment type descriptions were based on the Wentworth scale (Bale and Kenny, 2005).

2.5. Data analysis

Total number of individuals (N) (excluding colonial species) and total number of species (S) were calculated for each station. The significance of the difference between various groupings of stations, defined a priori (see Section 3.1.), was tested using one-way ANOVA.

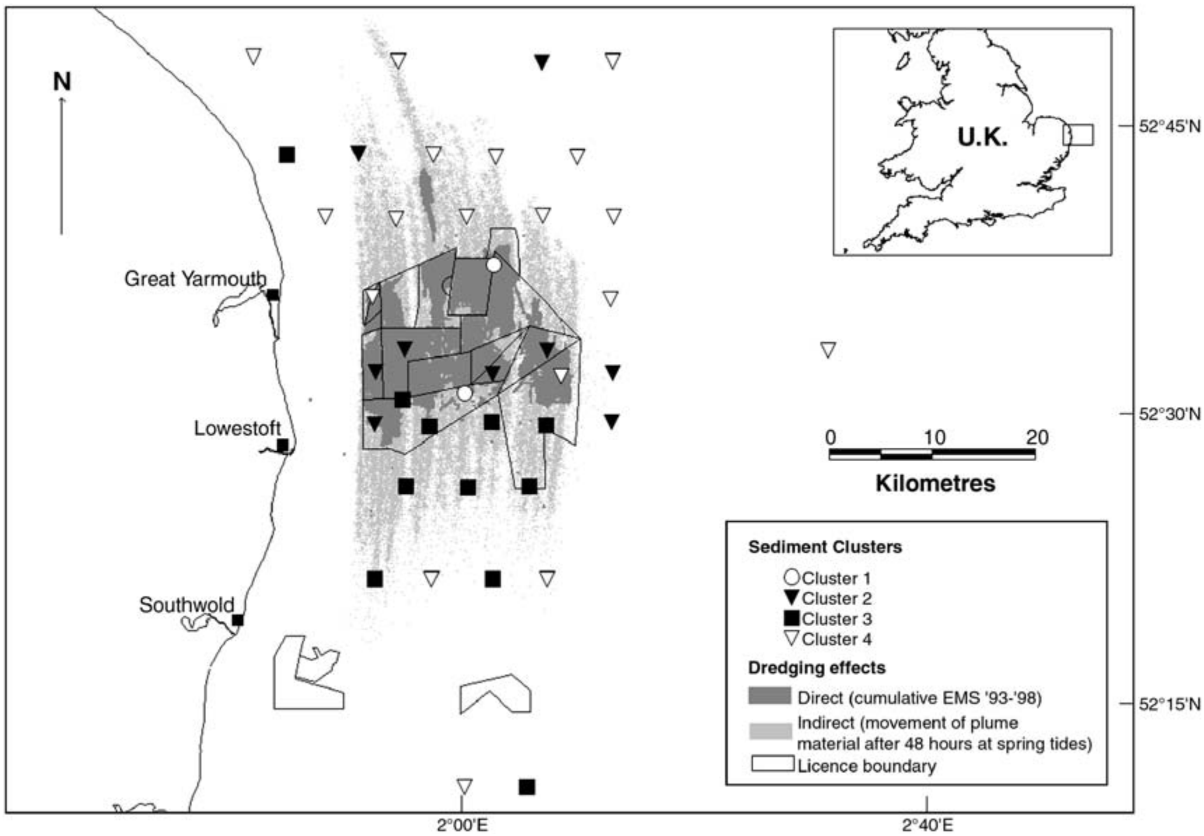


Fig. 4. The distribution of major sediment types based on cluster analysis of percentage particle size composition.

Where data failed to meet the necessary assumptions a non-parametric Kruskal–Wallis test (Kruskal and Wallis, 1952) was used. Fisher's Least Significant Difference (LSD) multiple comparisons procedure was used to determine the significance of differences between all sample groups. Univariate analyses were performed using the software package STATGRAPHICS Plus, Version 4.1.

Multivariate community analyses were carried out using the PRIMER® package (Clarke and Warwick, 1994). The data were fourth-root transformed to down-weight the importance of the very abundant species. Similarities were then calculated between every pair of samples, using the Bray–Curtis coefficient of similarity (Bray and Curtis, 1957) prior to group average clustering of the data (Lance and Williams, 1967) and production of multidimensional scaling ordination (MDS) plots. Following clustering, a series of 'similarity profile' (SIMPROF) permutation tests were used to look for statistically significant evidence of genuine clusters in the data. Analysis of similarities (ANOSIM) was used to test the significance of the difference between the different a priori groups of samples. The similarity percentages programme SIMPER was then used to identify the level of 'within-group' sample similarity, and also the species/sediment fractions responsible. The above analyses were also carried out on percentage particle size data. The significance of the relationship between similarity matrices underlying the macrobenthic and particle size datasets was tested using the RELATE test. Finally, in order to gain further insight into the distribution of macrofaunl communities, the BIOENV procedure was used to identify the environmental variables which best explained the observed patterns. This is achieved by selecting subsets of environmental variables which maximise the rank correlations between the two matrices. Variables included: % gravel, % coarse sand, % medium sand, % fine sand, % silt/clay, sorting, water depth (m), bed stress (N m^{-2}) and numbers of *Sabellaria spinulosa*. The latter was included as a variable due to its capacity to modify sediments locally and providing niches for a variety of other species.

3. Results

3.1. Extent of dredging

3.1.1. Direct impact

Maps showing the location and intensity of dredging (1993–1998) reveal that the annual area dredged has remained relatively consistent, varying between 80 km^2 and 110 km^2 (Fig. 1). However, over the same period the cumulative area dredged has increased from 90 km^2 in 1993 to 189 km^2 in 1998 (Fig. 1). Whilst this represents an approximate two-fold increase, the zone still only covers 57% of the total licensed area in 1998. However, it is important to appreciate that as dredging has taken place in this region since 1969, the 189 km^2 value may underestimate the true area dredged.

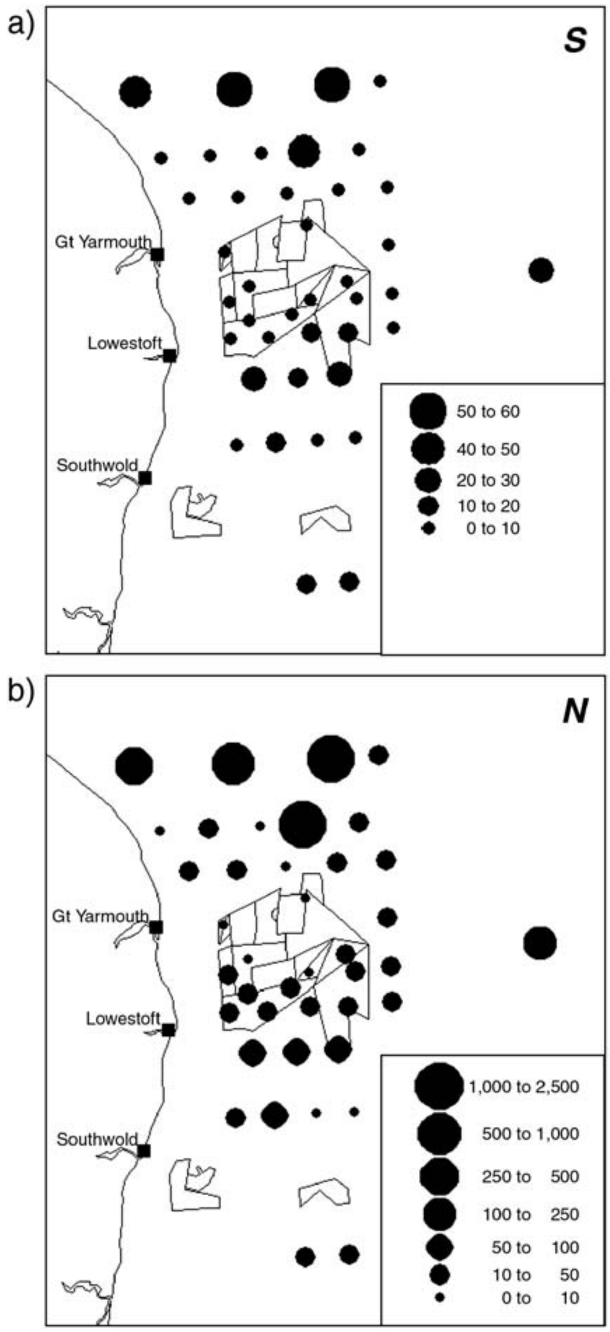


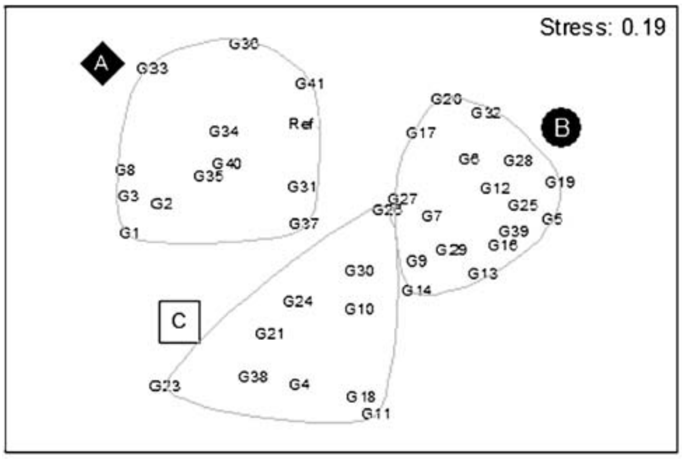
Fig. 5. Maps showing (a) number of species (S) and (b) number of individuals (N) found within the 40 0.1 m^2 Hamon grab samples taken across the survey area.

3.1.2. Indirect impact

The cumulative extent of seabed that may have been subject to 'indirect' effects of dredging is also shown in Fig. 2. This area was determined by modelling the movement of sediment plumes based on the known cumulative area dredged over the period 1993–1998. The zone occupies an area of approximately 700 km^2, nearly four times the size of the 'direct' impact zone, and extends approximately 12 km north and south of the 'direct' impact zone, well beyond the northern and southern limits of the licensed area. The zone extends slightly further on its western side as a result of stronger tides in this area. The lateral extent of the zone does not appear to go beyond the 'direct' zone, indicating very little potential for movement of material east and west of the licence area. As with the 'direct' zone, the fact that dredging has taken place in this region since 1969 means that the estimated area of 700 km^2 may be an underestimate of the true area subject to 'indirect' impacts resulting from dredging.

3.2. Regional assessment – sediments

The results of particle size analysis of sediment sub-samples from the broad-scale survey show that the survey area is dominated by sandy sediments (sand was the dominant fraction in 31 out of 40 stations sampled). However, more subtle differences in sediment composition were revealed by cluster analysis showing the separation of sediments into four groups (Fig. 3). Table 1 shows the characteristic composition of sediments from each cluster, identified using SIMPER analysis. With the



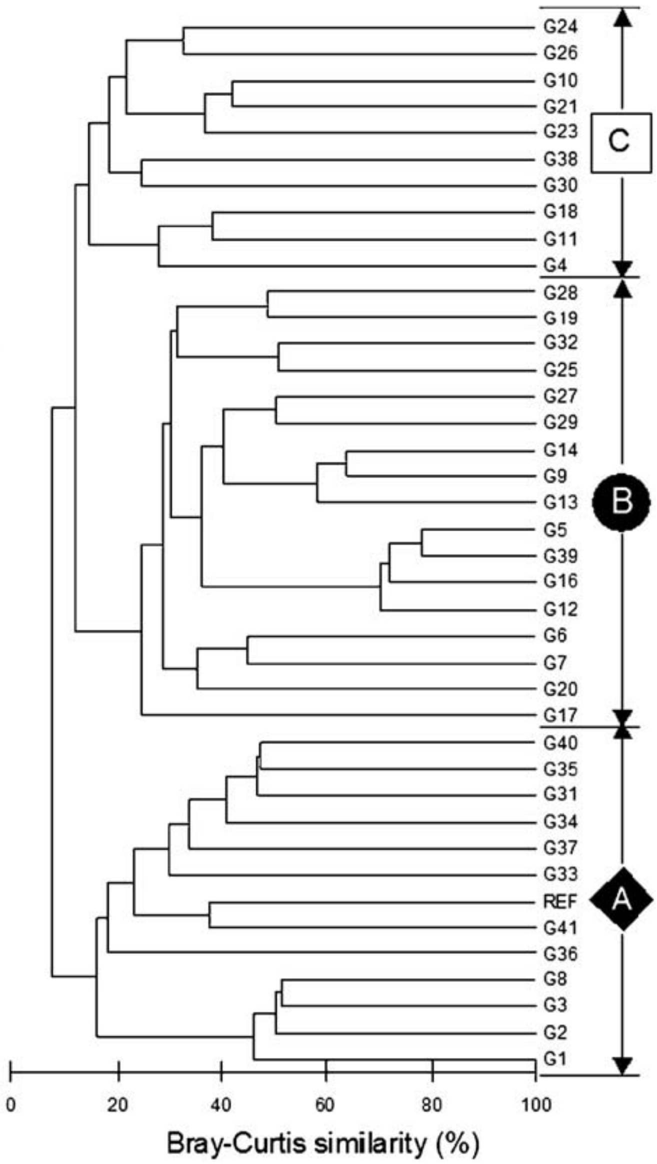


Fig. 6. Dendrogram showing clustering of samples based on macrofaunal composition.

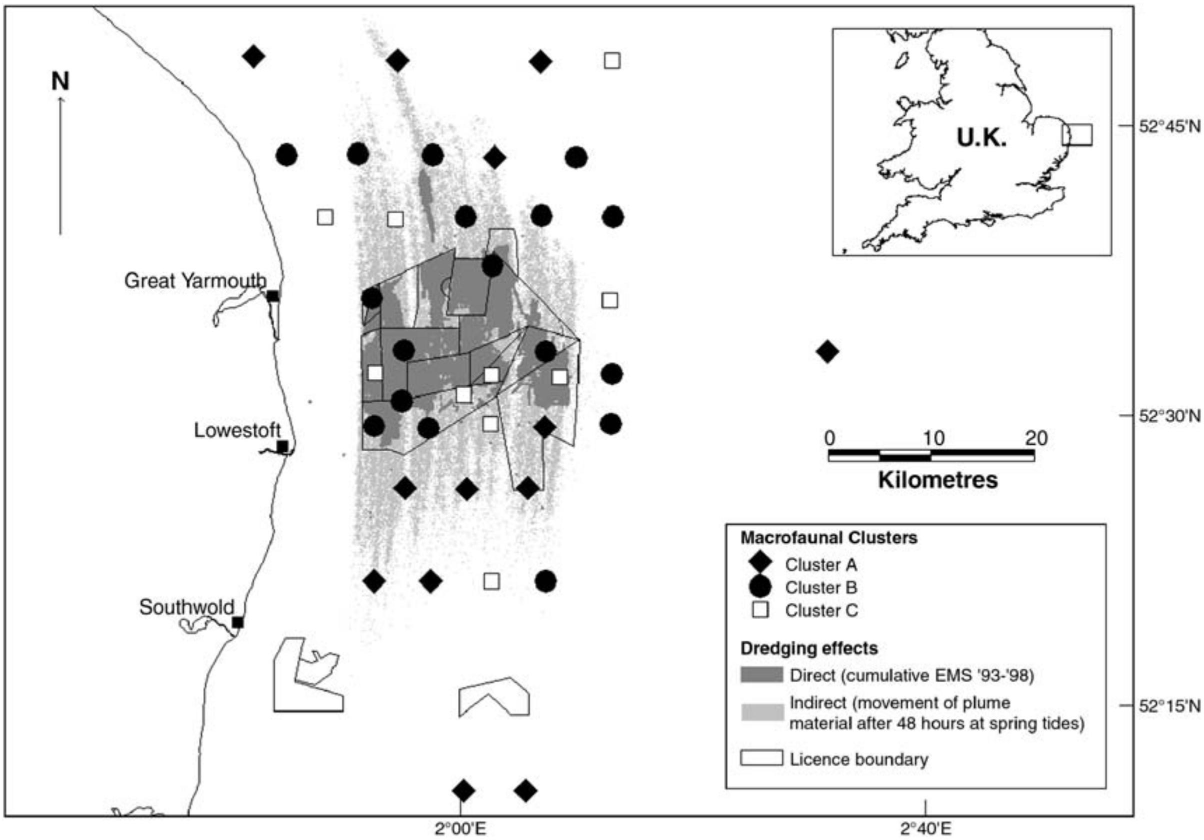


Fig. 7. The distribution of faunal assemblages identified from the cluster analysis of 4th root transformed abundance data from the 1998 broad-scale survey.

exception of group 1, dominated by silt/clay, all groups showed a high level of similarity between samples. Cluster 3 was dominated by gravel whilst clusters 2 and 4 were characterised by coarse and medium sands respectively. The distribution of these sediment clusters (Fig. 4) reveals a broad pattern of a fining of sediments moving northwards, with gravelly stations found in the southern half of the survey area, coarse sands in the central area and medium sands to the north. However, within this broad pattern there is some appreciable local variability.

3.3. Regional assessment– macrofauna

3.3.1. Univariate

A total of 192 taxa were identified from the 40 Hamon grab samples. Fig. 5, showing the number of species and individuals, clearly illustrates the sparse nature of the fauna in the region, although some relative 'hot spots' clearly exist. All of the species found were typical of mobile sandy sediments. The numbers of infaunal species found within individual grab samples ranged from 2 to 61 and total densities ranged from 3 to

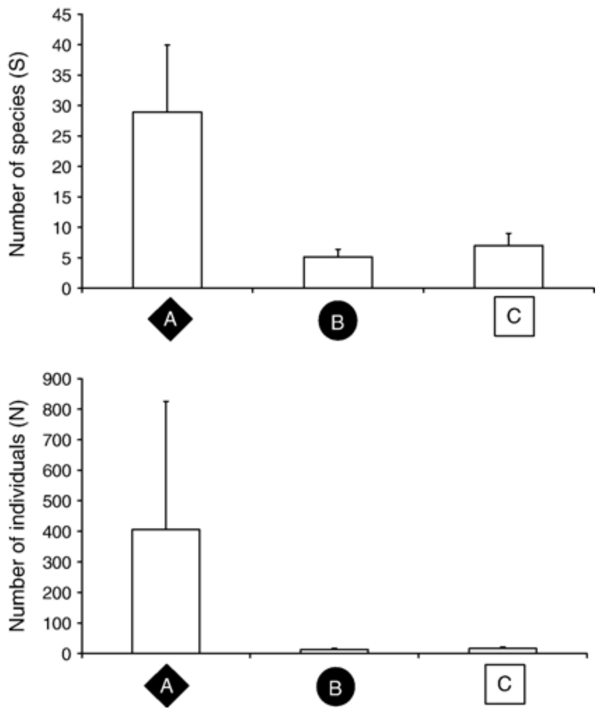


Fig. 8. Summary of means and 95% confidence intervals for number of species (S) and individuals (N) from each macrofaunal assemblage identified in the cluster analysis.

2405 individuals per 0.1 m^2. *Ophelia borealis* was the most common species and was found at 63% of the stations sampled.

3.3.2. Multivariate

Overall, levels of community similarity between samples were very low. However, cluster analysis revealed three groups: A, B and C at the 14% level of similarity (see Fig. 6). SIMPROF tests confirmed A as being statistically significantly distinct, but clusters B and C could not be significantly differentiated. Assemblage A was found at 13 stations in the northern, southern and eastern extremes of the survey area whilst B and C were found within, to the north, and immediately to the east of the licensed areas (Fig. 7). Assemblage B was found at 17 stations and assemblage C at 10. All 3 clusters were found in all 3 zones with the exception of A which was absent from the 'direct' zone.

A comparison of the mean values of number of species and individuals found within each cluster (Fig. 8) revealed low values of both measures within clusters B and C. In contrast, values of both measures

Table 2
Results from SIMPER analysis of Hamon grab data (Colonials excluded, 4th root transformed), listing the main characterising species within each assemblage type

Cluster	Taxonomic group	Average abundance	Average similarity	% Contribution	Cumulative %	Average similarity %
◆	*Lanice conchilega*	8.3	3.6	15.2	15.2	23.9
	NEMERTEA	6.0	1.9	7.9	23.1	
	Sabellaria spinulosa	181.1	1.8	7.7	30.8	
	Spiophanes bombyx	2.5	1.4	5.9	36.7	
	Ophiura albida	4.8	1.2	5.2	41.9	
	Scoloplos armiger	3.2	1.2	4.9	46.8	
	CNIDARIA	9.2	1.1	4.5	51.2	
	Ophiura sp.	7.1	0.9	3.8	55.0	
	Aonides paucibranchiata	0.7	0.9	3.6	58.5	
	Mysella bidentata	2.8	0.8	3.3	61.8	
	pholoe inornata	4.5	0.8	3.1	64.9	
	Polycirrus sp.	6.5	0.7	2.7	67.7	
	Marphysa bellii	0.5	0.6	2.6	70.3	
	Echinocyamus pusillus	0.4	0.6	2.5	72.8	
	Keferteinia cirrata	2.1	0.4	1.7	74.5	
	Urothoe elegans	8.5	0.4	1.7	76.1	
	Abra alba	6.8	0.4	1.5	77.6	
	Ampelisca spinipes	1.2	0.3	1.4	79.0	
	Goniada maculata	0.6	0.3	1.4	80.4	
	Scalibregma inflatum	0.9	0.3	1.1	81.5	
	Caulleriella alata	1.2	0.3	1.1	82.7	
	Harmotoe impar	4.2	0.2	1.0	83.6	
	Glycera lapidum	2.2	0.2	1.0	84.6	
	Owenia fusiformis	0.4	0.2	0.9	85.5	
	Ophelia borealis	0.8	0.2	0.9	86.5	
	Pisidia longicornis	68.6	0.2	0.9	87.4	
	Eumida bahusiensis	0.9	0.2	0.9	88.3	
	Amphipholis squamata	8.6	0.2	0.9	89.1	
	Nephtys cirrosa	0.5	0.2	0.8	89.9	
	Aora gracilis	1.1	0.2	0.8	90.7	
●	*Ophelia borealis*	7.2	25.5	75.7	75.7	33.7
	Glycera oxycephala	0.4	2.4	7.2	82.9	
	Nephtys cirrosa	0.6	1.7	5.0	87.9	
	Urothoe brevicornis	0.5	0.8	2.4	90.2	
C	*Scoloplos armiger*	2.0	9.3	47.3	47.3	19.7
	Sabellaria Spinulosa	2.0	3.2	16.5	63.8	
	Magelona mirabilis	0.6	1.8	8.9	72.7	
	Glycera lapidum	0.5	1.7	8.6	81.3	
	Urothoe brevicornis	2.0	0.9	4.5	85.9	
	Nephtys cirrosa	0.5	0.8	4.1	90.0	
	Ophelia borealis	1.1	0.8	3.8	93.8	

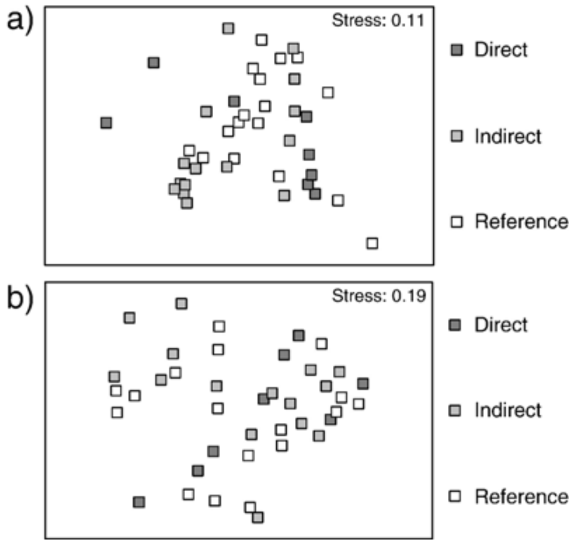


Fig. 9. Multidimensional scaling (MDS) ordination of (a) percentage particle size distributions (untransformed data) and (b) macrobenthic data (fourth-root transformed). Samples are colour coded according to their position in relation to the location of dredgeing impacts (see Fig. 1).

within cluster A were higher. Samples within cluster A also showed the highest occurrence of colonial epifaunal animals. No formal significance tests were applied to these clusters as they were identified post priori.

Clusters also differed in terms of the mean percentage composition of major taxonomic groups, with polychaetes making up a higher proportion of the fauna within assemblages B and C, both in terms of number of species and individuals.

The results of a SIMPER analysis to identify the characterising species from each cluster (Table 2) clearly show the much-reduced range of species that characterised clusters B and C. However, the majority of species found in clusters B and C were also found in A. Polychaetes were the most important group in all clusters although tube-dwelling sedentary species such as *Lanice conchilega*, *Sabellaria spinulosa* and *Spiophanes bombyx* dominate assemblage A, whilst more errant species such as *Ophelia borealis* and *Glycera* sp. dominated the fauna in groups B and C. All species encountered are typical of high-energy sandy environments.

3.4. Assessment of dredging impact

Sampling stations were assigned to either the 'direct', 'indirect' or 'reference' category depending on their position in relation to predicted impacts from the outcome of modelling exercises (see Fig. 2). These groupings were then used to determine whether there was any difference between the sediments and fauna found in these zones which might provide evidence of a broad-scale cumulative impact resulting from dredging.

3.4.1. Sediments

No distinction between impact zones was identified from an MDS ordination for particle size data (Fig. 9a). However, ANOSIM identified a significant difference between the 'direct' and 'indirect' zone ($R=0.249$, $p=0.01$). SIMPER analysis revealed that this difference was mainly a result of less gravel within the 'direct' zone and proportionally more coarse sand.

3.4.2. Macrofauna

An MDS ordination for the macrofauna data showed no significant difference between sample groups from the three dredging impact zones (Fig. 9b), and this was confirmed by an ANOSIM test ($R=-0.015$, $P=0.59$). In contrast, plots of the mean numbers of species (S) and individuals (N) (Fig. 10) suggest a negative correlation between the status of the benthic fauna and the severity of impact. A multiple comparisons procedure revealed significant differences ($p<0.05$) between 'direct' and 'reference' conditions for a number of species. Similarly, a Kruskal wallis test revealed statistically significant differences between the median values of number of individuals between 'direct' and 'reference' areas ($p=0.036$). Further analysis revealed that this pattern

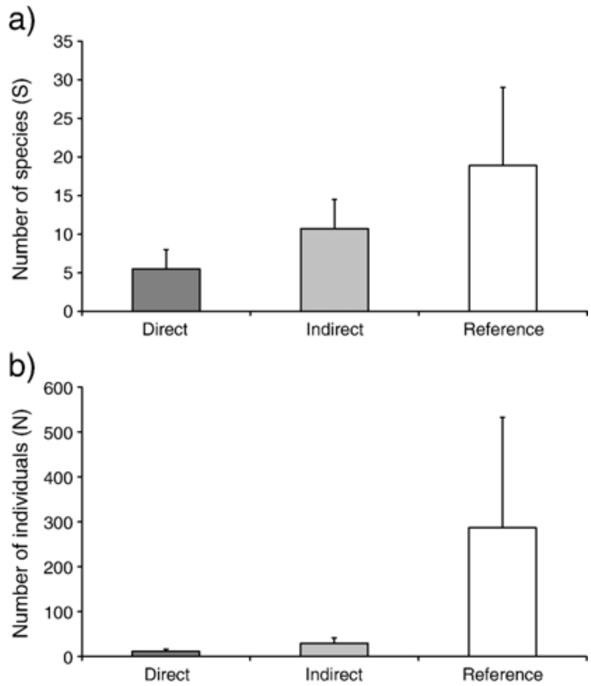


Fig. 10. Summary of means and 95% confidence intervals for number of species (S) and individuals (N) found in samples taken from the dredging impact areas identified in Fig. 2.

was evident across all major taxonomic groups (polychaetes, crustaceans, molluscs, echinoderms and 'others').

3.5. Further exploration of factors affecting faunal distributions

3.5.1. Sediments

Results from the RELATE procedure indicated no relationship between the biotic and particle size datasets (Rho = 0.013, p > 0.05). This may suggest that the overall composition of sediments is not the over-riding factor responsible for the distribution of communities in this region.

In order to investigate whether this observation was true for all taxa, individual species abundances were overlaid on the multidimensional scaling ordination derived from the sediment particle size data (see Fig. 11). Most species showed no apparent preference for any of the sediment groups (e.g. *Ophelia borealis*). However, *Nephtys caeca* was only found within the gravel-rich sediments from cluster 3 and *Magelona mirabilis* was only found on the predominately medium sands in cluster 4. Interestingly, the results indicate that certain species may exhibit avoidance and preference in relation to the predominately coarse sands in cluster 2. Those species showing avoidance include: *Bathyporeia* spp., *Nephtys cirrosa, Echinocyamus pusillus, Ampelisca spinipes* and *Urothoe* spp. In contrast, *Notomastus* spp. and *Eurydice spinigera* were only found within the sediments from this cluster.

3.5.2. All factors

The results of a BIOENV analysis revealed the best fit between the macrofauna community patterns and a single environmental variable was achieved with

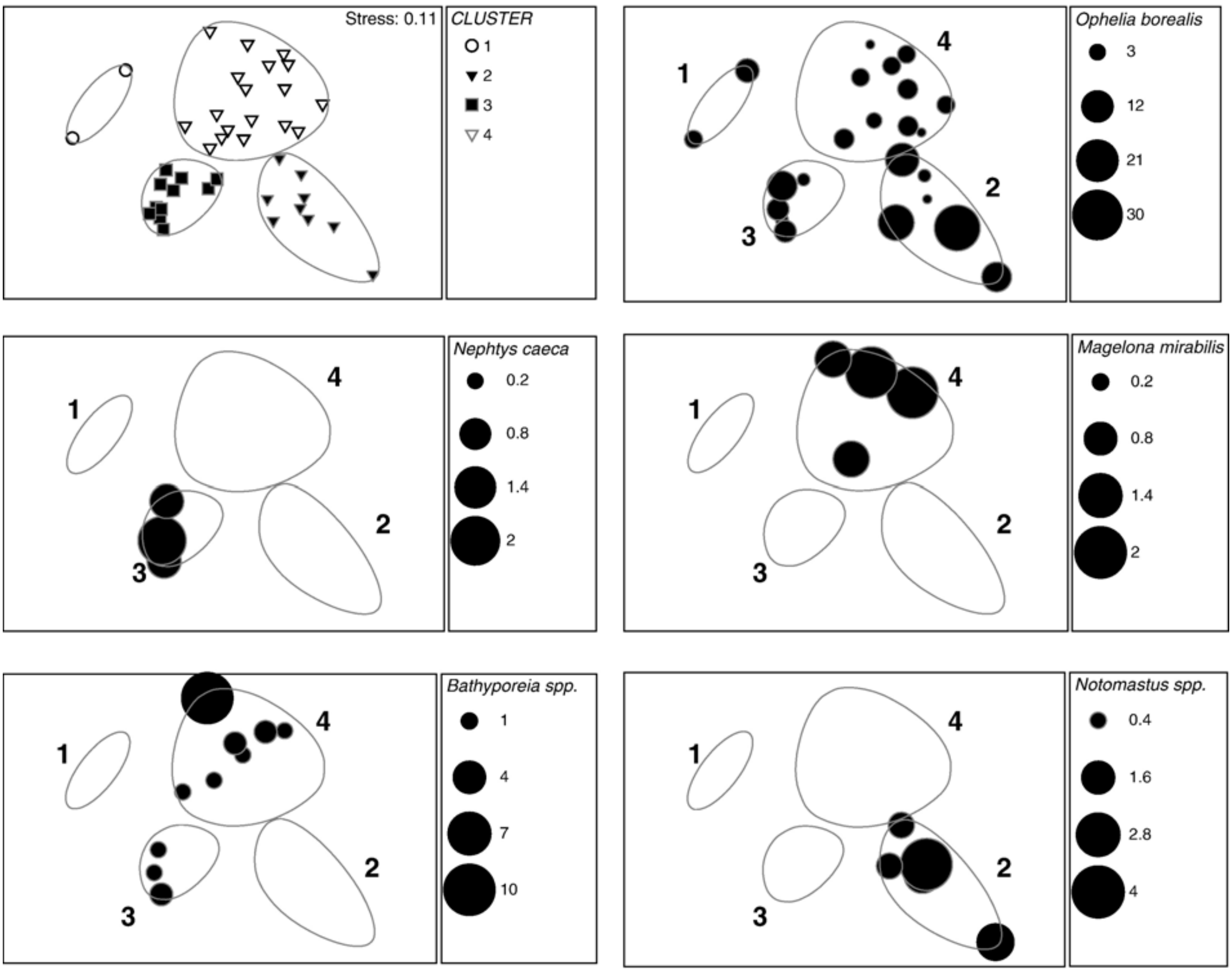


Fig. 11. Faunal abundances overlaid on the particle size MDS ordination.

numbers of *Sabellaria spinulosa*. The fit was improved with the addition of a further two variables (coarse sand and sorting) to give the highest correlation of 0.306.

4. Discussion

Overall, the predominately sandy sediments found across much of the region supported sparse macrofaunal assemblages. This is in agreement with previous work carried out in the area by Kenny et al. (1991) and Rees et al. (1999). These authors attributed the sparse nature of the fauna to a combination of tidally-induced sediment mobility and the abrasive effects of sand in suspension, inhibiting the development of many epifaunal species on any exposed gravelly components. This is supported by the observations of Millner et al. (1977) and HR Wallingford (1995) that the strength of tidal currents found in the area are more than sufficient to mobilise medium sands. Another consequence of this mobility is the likelihood of a rapid infilling of dredge tracks (Eden, 1975; Kenny et al., 1991), in contrast to more stable areas where these features can persist for several years after cessation of dredging (Millner et al., 1977; Dickson and Lee, 1973a,b; Boyd et al., 2003, 2005; Diesing et al., in press).

In high-energy environments, soft sediment faunal assemblages typically occupy an early successional stage (Newell et al., 1998; Bolam and Rees, 2003) characterised by the dominance of species displaying ‘r-selected’ characteristics, namely rapid reproduction, short life span and high dispersal potential (Kröncke, 1990; Niermann et al., 1990). As a result these assemblages have the potential to recover quickly following super-imposed disturbance, such as may arise from dredging or disposal, in contrast to more stable environments.

Numbers of *Sabellaria spinulosa*, sorting coefficient and percentage coarse sand were found to be the most influencial variables that accounted for patterns in the faunal data. As sorting can be a useful expression of the dynamic nature of the local physical environment (Rees et al., 1999) these results suggest, in common with other studies (Cabioch, 1968; Warwick and Uncles, 1980; Holme and Wilson, 1985; Kenny et al., 1991), that tidal current velocity and sediment characteristics are important as causal agents in explaining variability between stations. The faunistically richer areas identified in this study may be due to marginally enhanced sediment stability arising from the higher proportion of gravel found in samples to the south of the extraction licences and to the presence of *S. spinulosa* reef in the north. The occurrence of the latter reef is known to provide niches for a variety of other species (Foster-Smith and White, 2001).

Further insight into faunal distributions in this region is provided by Kenny et al. (1991) who observed a superficial homogenous habitat of sand at an extensively dredged site off Lowestoft. The mobility of this material, as evidenced by sand waves and ripples from sidescan sonar data, appeared to explain why only a few dredge tracks were visible. These observations suggest that, although quantities of gravel may be present in the well-mixed Hamon grab samples from the present study, it may not always be present at the seabed surface, and therefore available for colonisation. This observation may further help to explain the absence of a strong correlation between sediment particle size and macrofaunal assemblage compositions in this study.

The comparison of particle size data from dredging impact zones revealed that sediments from within the ‘direct’ area contained slightly less gravel, and proportionally more sand than sediments from the ‘indirect’ zone. This provides evidence of a possible near-field effect of dredging on sediments. Evidence from a number of studies shows dredging can lead to the ‘fining’ of sediments, particularly where screened material is returned to the seabed (Wallingford, 1995; Robinson et al., 2005). This can lead to interference of feeding activity or even the burial of the fauna, the significance of which will depend on several factors including the rate of sedimentation, sediment type, and the ability of benthic organisms to cope with a rapid accretion of sediment (Maurer et al., 1981a,b, 1982; Schratzberger et al., 2000). In addition, in ‘high-energy’ areas the reduction in the quantity of the gravel may have the effect of reducing sediment stability and consequently restricting re-colonisation potential, other than by a few very tolerant species. Evidence of this is provided by Desprez (2000) off the French coast, who noted that an *Ophelia* community, naturally present to the east of the extraction area, appeared in dredged areas due to a proliferation of mobile sands. Higher proportions of sand, in areas exposed to the highest levels of dredging intensity, were also observed by Boyd et al. (2003, 2005) at a licensed extraction site in the southern North Sea.

Whilst multivariate analyses of the macrofaunal data were unable to discriminate between ‘direct’, ‘indirect’ and ‘reference’ zones, plots of mean numbers of species (S) and individuals (N) showed statistically significant differences between the ‘direct’ and ‘reference’ areas. This provides good evidence of the near-field consequences of dredging. However, since there was an element of targeting of stations in areas of known current dredging activity, it would be unreasonable to conclude that the entire zone is uniformly impoverished as a result of the cumulative consequences of dredging. Of potentially greater interest is the possibility that dredging-related effects may extend beyond licensed boundaries as a result of the re-distribution of finer material disturbed or discharged during the dredging process. In this respect, values of both univariate measures within the 'indirect' zone were lower than in the 'reference' zone, but these differences were not statistically significant. We conclude that, although the dominant influence on assemblages in the region is that of tidally-induced sediment instability, we cannot dismiss the possibility of a subsidiary influence of dredging activity in the near vicinity of the licensed block and further investigation is warranted.

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Paper #12

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The effects of marine sand and gravel extraction on the macrobenthos at a commercial dredging site (results 6 years post-dredging)

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Boyd, S. E., Limpenny, D. S., Rees, H. L., and Cooper, K. M. 2005. The effects of marine sand and gravel extraction on the macrobenthos at a commercial dredging site (results 6 years post-dredging). – ICES Journal of Marine Science, 62: 145–162.

Benthic recolonization was investigated at a site historically used for the extraction of marine sand and gravel. The main objective was to assess the effects of different levels of dredging intensity on the recolonization of benthic fauna and sediments. Preliminary observations from this study indicated that the fauna within an area of seabed exposed to high dredging intensities remained in a perturbed state some 4 years after the cessation of dredging. Thereafter, annual monitoring surveys of the benthos and sediments at the "treatment" and "reference" sites have followed the recolonization process. Results from univariate and multivariate data analyses show that distinct differences in the nature of assemblages at sites exposed to high and lower levels of dredging intensity persist at least 6 years after the cessation of dredging. This paper presents the physical and biological findings 6 years after dredging, together with a generic framework for evaluating post-cessation recolonization studies.

Keywords: aggregate extraction, dredging, impacts, North Sea, recolonization.

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Introduction

Much of the seabed surface around the England and Wales coastline consists of coarse material, i.e. various proportions of sand and gravel (CIRIA, 1996). Where these resources are present in sufficient quantity, are of the right composition, and are accessible to commercial dredgers, they may be exploited as a source of aggregate for the construction industry, to supplement land-based sources, or as a source of material for beach nourishment (Singleton, 2001).

As the extraction of marine sand and gravel has its primary impact at the seabed, assessment of the effects of this activity has conventionally targeted bottom substrata and the associated benthic fauna (Millner *et al.*, 1977; Desprez, 2000; Van Dalfsen *et al.*, 2000). Historically, the scientific study of coarser substrata has presented a significant challenge, largely on account of the difficulties of obtaining reliable quantitative samples (Eleftheriou and Holme, 1984). As a consequence, information on the nature and distribution of benthic assemblages, and on their wider role in the marine ecosystem, is considerably more limited than in areas of soft sediments.

Of those studies which have considered the effects of marine aggregate extraction, most have concentrated on establishing the rates and processes of macrobenthic recolonization upon cessation of dredging (Cressard, 1975; Kenny *et al.*, 1998; Desprez, 2000; Sardá *et al.*, 2000; Van Dalfsen *et al.*, 2000; Van Dalfsen and Essink, 2001). These studies indicate, typically, that dredging causes an initial reduction in the abundance, species diversity, and biomass of the benthic community (for review see Newell *et al.*, 1998) and that substantial progress towards full restoration of the fauna and sediments can be expected within a period of approximately 2–4 years following cessation (Kenny *et al.*, 1998; Sardá *et al.*, 2000; Van Dalfsen *et al.*, 2000; Van Dalfsen and Essink, 2001). For example, Van Dalfsen *et al.* (2000) suggested that recolonization of a dredged area by polychaetes occurred within 5–10 months after the cessation of dredging in a site located within the North Sea, with restoration of biomass to pre-dredge levels anticipated within 2–4 years. Such studies have been mainly concerned with the effects of dredging operations conducted over a relatively short time scale, e.g. up to periods of 1 year (Kenny *et al.*, 1998; Sardá *et al.*, 2000; Van Dalfsen *et al.*, 2000; Van Dalfsen and

Essink, 2001). Under such circumstances, any more subtle effects, e.g. on seasonal recruitment success to the locality, arising from prolonged dredging over several years would clearly be expected to be minimal. Few studies have addressed the consequences of long-term dredging operations (Desprez, 2000). Thus, there is limited information which is directly applicable to the impacts of commercial dredging operations in UK waters where the life-time of a typical production licence is at least 15 years. The aim of this study was to assess the status of the seabed substrata and associated benthic assemblages at a former extraction site which was intensively dredged over a 25-year period. Preliminary observations on the status of this extraction site, 4 years after the cessation of dredging, were reported in Boyd *et al.* (2003). In this paper we examine the findings from surveys carried out between 2000, and 2002 i.e. 4, 5, and 6 years on, and investigate whether different historical levels of dredging intensity affect the subsequent rate and nature of benthic recolonization at an aggregate extraction site following cessation of dredging.

Methods

Study site

A full description of the study site (designated "Area 222") together with an account of the dredging history is reported in Boyd *et al.* (2003). It is located approximately 20 miles east of Felixstowe off the southeast coast of England (Figure 1) in water depths of between 27-m and 35-m Lowest Astronomical Tide (LAT). The tidal ellipse in the region is rectilinear and is aligned in a NNE-SSW direction which is thought to be modified by an adjacent deeper channel that encroaches into the northern edge of the extraction site. Maximum spring tidal current velocities reach 1.5 m s^{-1} and there is evidence for a NNE nearbed residual tidal direction (Boyd *et al.*, 2003). This site, with dimensions of approximately 900 m by 300 m, was first licensed for sand and gravel extraction in 1971, with a peak in extraction activity recorded as 872 000 t in 1974. Extraction continued at levels $>100\,000$ t per annum until 1995, before the site was relinquished by the industry in 1996. At this site, the sand:gravel ratios of dredged cargoes were adjusted by screening, with excess sand being discharged overboard at the site of dredging.

Sampling design

Since 1993, every vessel dredging on a Crown Estate licence in the UK has been fitted with an Electronic Monitoring System (EMS) which automatically records the date, time, and position of all dredging activity, every 30 s, to disk. EMS information was interrogated in order to locate areas of the seabed within the Area 222 extraction licence which had been subjected to different levels of dredging intensity. Replicate samples of the macrofauna and sediments were collected from areas representing two different levels as follows (i) >10 h of dredging within a 100-m by 100-m block during 1995 and (ii) <1 h of dredging within a 100-m by 100-m block during 1995. EMS data from 1995 were chosen since this year was the last year that the licensed extraction site was dredged

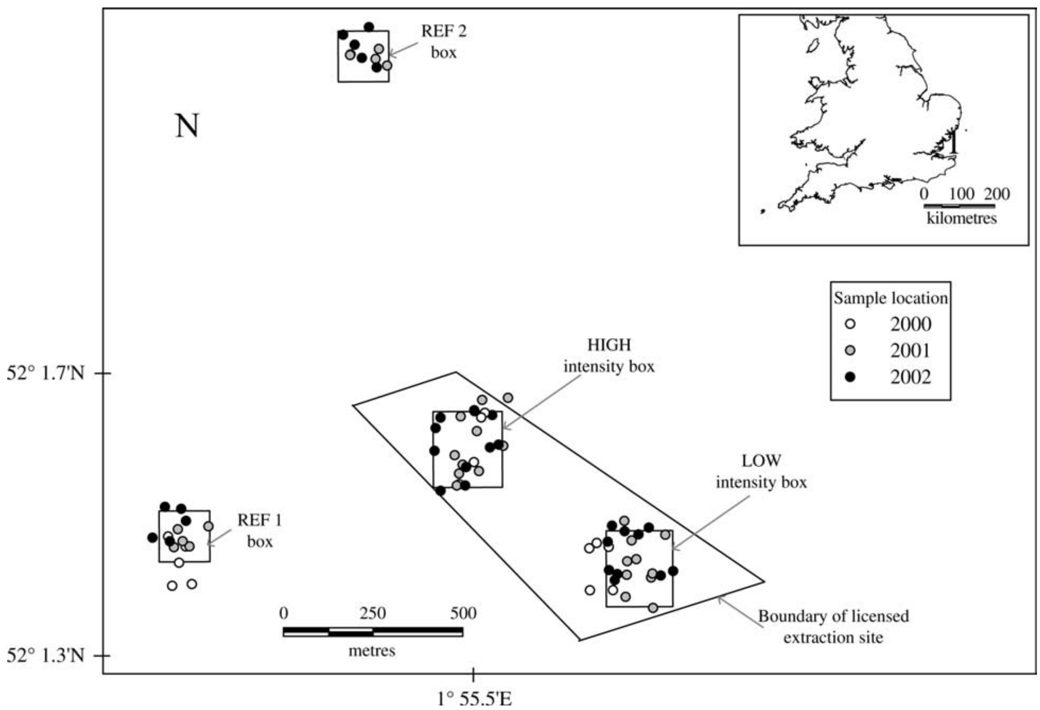

Figure 1. Map showing the location of the Area 222 extraction licence and sample locations from 2000 to 2002 in the southern North Sea.

heavily. The location and intensity of dredging was comparable between 1993 and 1995. In addition, a reference site (Reference site 1) was sampled in 2000–2002 and this was augmented by sampling at a second reference site (Reference site 2) in 2001 and 2002. Boyd *et al.* (2003) present an account of the design in terms of the likely dredging impact. Stations were randomly distributed within each area ("stratified random sampling") and allocated in proportion to the size of the sampling box (Green, 1979). Reference sites were selected as being representative of the wider environment surrounding the extraction site and outside the influence of any potential effects on the benthos from dredging (see Figure 1). Selection of appropriate reference sites was aided by the use of sidescan sonar and video images of the seabed (see Boyd *et al.*, 2003 for methodology) and following criteria given in CSTT guidelines (1997) and Boyd (2002). There was also no evidence of the effects of other forms of seabed disturbance at the reference locations, i.e. effects of trawling activity. Data arising from this design provide a comparative evaluation of "treatment" and "reference" groups (e.g. Skalski and McKenzie, 1983). Note that the "reference" areas are not necessarily representative of baseline conditions, as there is insufficient historical information on which to determine what actually constitutes the likely pre-dredging status of Area 222.

Area 222 was not dredged in the 4 years prior to sampling. Sampling was conducted in July 2000, 2001, and 2002, that is 4, 5, and 6 years after the cessation of dredging. Sampling details for locations sampled as part of this study are presented in Table 1.

Sample collection

Samples for analysis of the macrobenthic fauna and sediment particle size were collected with a 0.1-m^2 Hamon grab from RV "Cirolana". This device was employed because it has been shown to be particularly effective on coarse substrata (Kenny and Rees, 1994, 1996; Seiderer and Newell, 1999).

Following estimation of sample volume, a 500-ml subsample was removed for laboratory sediment particle size analysis. The whole sample was then washed over 5-mm and 1-mm square mesh sieves to remove the fine sediment. The two resultant fractions (1–5-mm and >5-mm) were back-washed into separate containers and fixed in 4–6% buffered formaldehyde solution (diluted in seawater) with the addition of "Rose Bengal", a vital stain.

Macrofauna samples were processed according to the guidelines given in Boyd (2002). The >5-mm sample fraction was first washed with freshwater over a 1-mm mesh sieve in a fume cupboard, to remove excess formaldehyde solution, then back-washed onto a plastic sorting tray. Specimens were removed and identified, where possible, to species level. The 1-5-mm fraction was first washed over a 1-mm sieve then back-washed into a 10-litre bucket. The bucket was filled with freshwater and the sample was then gently stirred in order to separate the animals from the sediment. Once the animals were in suspension, the sample was decanted over a 1-mm mesh sieve. This process was repeated until no more material was recovered. Specimens from this fraction were placed into labelled petri dishes for identification and enumeration. The sediment was then placed on plastic trays and examined under an illuminated magnifier for any remaining animals such as bivalves not recovered in the decanting process, which were then added to the petri dishes.

Sediment particle size analysis

The sediment subsamples from each grab were analysed for their particle size distributions. Samples were first wet-sieved on a 500-µm stainless steel test sieve using a sieve shaker. The <500-µm sediment fraction passing through the sieve, was allowed to settle from suspension in a container for 48 h. The supernatant was then removed

Table 1. Sampling details for locations sampled as part of the time-series investigations at Area 222. Box coordinates given as positions in WGS 84 from top right and bottom left hand corners of the sampling box.

		Box coordinates			Number of samples collected		
Treatment	Code	Longitude	Latitude	Area (m^2)	2000	2001	2002
High intensity box	HIGH '00 to '02	52° 01.686′N 52° 01.572′N	01° 55.554′E 01° 55.386′E	40 000	5	10	10
Low intensity box	LOW '00 to '02	52° 01.506′N 52° 01.392′N	01° 55.968′E 01° 55.806′E	40 000	5	10	10
Reference site 1	REF1 '00 to '02	52° 01.530′N 52° 01.470′N	01° 54.828′E 01° 54.726′E	20 000	5	5	5
Reference site 2	REF2 '01 to '02	52° 02.256′N 52° 02.184′N	01° 55.278′E 01° 55.158′E	20 000	0	5	5

using a vacuum pump and the remaining <500-μm sediment fraction was washed into a petri dish, frozen for 12 h, and freeze-dried. The total weight of the freeze-dried fraction was recorded. A subsample of the <500-μm fraction was then analysed using a laser sizer and a percentage weight for each size class was calculated. The >500-μm fraction was washed from the test sieve into a foil tray and oven dried at ~90°C for 24 h. It was then dry sieved on a range of stainless steel test sieves, placed at 0.5-phi intervals, down to 1 phi (500 μm). The sediment on each sieve was weighed to 0.01 g and the values recorded. The results from these analyses were combined to give a full particle size distribution for each sample.

Acoustic surveys

Sidescan sonar surveys were undertaken using the Datasonics™ SIS 1500 digital chirps system using the Triton Isis™ data acquisition software. The Delphmap™ software package was used to post-process the data, and provided georeferenced mosaic images of the sonar data. Such surveys were undertaken in order to provide an indication of the spatial distribution of sediments in the wider area encompassing the dredged sites and to provide information on the distribution and stability of bedforms. Such information contributes to an evaluation of *inter alia* the physical "recovery" of sites, e.g. the persistence of dredged tracks or pits.

Data analysis

Sediment variables

Particle size distribution data have been presented using cumulative frequency distribution curves. Changes in the shape of the curve for any given sample when compared to another, reflect the variations in the particle size distribution of those samples.

A correlation-based principal components analysis (PCA) was applied to ordinate results from the sediment analyses (Clarke and Warwick, 1994).

Analysis of similarities (ANOSIM, Clarke, 1993) was performed on sediment particle size data to test the significance of differences in particle size composition between treatments.

Macrofaunal assemblage structure

The total numbers of individual organisms and numbers of species were calculated for each sample group. This allows a visual interpretation of any trends (e.g. increasing or decreasing abundance at different sampling locations and over time) and their statistical significance, whereas this judgement is more difficult for results obtained by multivariate data analyses. The significance of differences between treatments was tested using one-way ANOVA.

A non-parametric multi-dimensional scaling (MDS) ordination using the Bray–Curtis similarity measure (Bray and Curtis, 1957) was applied to species abundance data.

Warwick and Clarke (1993) noted that in a variety of environmental impact studies, the variability among samples collected from impacted areas was much greater than that collected from reference sites. They suggested that this variability was in itself an identifiable symptom of perturbed situations. To test whether this pattern was evident with the data from dredged sites examined in this study, the comparative Index of Multivariate Dispersion (IMD) was calculated. IMD has a maximum value of +1 when all similarities among impacted samples are lower than any similarities among reference samples. The converse case gives a minimum for IMD of −1, and values near zero imply no difference between groups.

The comparative Index of Multivariate Dispersion is restricted to the comparison of only two groups, e.g. reference vs. high dredging intensity samples and therefore is usually complemented by calculation of the relative Index of Multivariate Dispersion (r.IMD; Somerfield and Clarke, 1997). This index has a value of 1 if the relative dispersion of samples corresponds to the "average dispersion". Values greater than 1 are obtained if replicate samples are more variable than average. In contrast, a value lower than 1 is achieved if replicate samples are less variable than average.

Analysis of similarities (ANOSIM, Clarke, 1993) was performed to test the significance of differences in macrofauna assemblage composition between samples. The nature of the groupings identified in the MDS ordinations was explored further by applying the similarity percentages program (SIMPER) to determine the contribution of individual species to the average dissimilarity between samples.

All multivariate analyses were performed using the software package PRIMER v. 5, developed at the Plymouth Marine Laboratory (Clarke and Gorley, 2001).

Results

Sediment characteristics

Sediment particle size characteristics are presented in Table 2 and the cumulative particle size distribution curves for each of the survey years are presented in Figure 2. Grain size descriptions relate to the Udden–Wentworth scale (Wentworth, 1922). Particle size data revealed that there was a high degree of variability between replicate samples, particularly in the gravel and sand components of the distributions.

In both 2000 and 2001, the replicate sediments sampled at the site of high dredging intensity show a large degree of variability in the gravel and coarse sand fractions. In 2002, an apparent reduction of the gravel component at this

Table 2. Mean values (±s.d.) of sediment particle size characteristics for each treatment (codes as in Table 1).

Year	Treatment	Mean particle size [mm]	Sorting	Skewness	Kurtosis	Gravel [%]	Coarse sand [%]	Medium sand [%]	Fine sand [%]	Silt/clay [%]
2002	REF1 02	0.54 (±0.36)	4.71 (±0.55)	0.28 (±0.29)	1.77 (±0.36)	40.78 (±7.97)	12.16 (±2.74)	7.85 (±5.65)	4.92 (±1.36)	34.30 (±10.91)
	REF2 02	2.06 (±1.56)	3.63 (±0.97)	0.82 (±0.77)	3.57 (±1.36)	51.42 (±15.59)	19.61 (±8.73)	6.00 (±3.43)	4.11 (±0.70)	18.85 (±23.64)
	LOW 02	1.23 (±0.72)	3.44 (±0.57)	0.58 (±0.25)	2.97 (±0.72)	44.35 (±9.81)	10.34 (±4.21)	23.48 (±6.66)	10.57 (±3.193)	11.27 (±7.12)
	HIGH 02	1.29 (±0.69)	1.94 (±0.96)	0.02 (±0.68)	6.06 (±4.10)	27.24 (±15.86)	49.86 (±20.91)	18.69 (±6.28)	1.81 (±1.73)	2.40 (±5.25)
2001	REF1 01	1.04 (±0.65)	3.81 (±0.68)	0.76 (±0.21)	3.08 (±1.16)	43.72 (±13.54)	20.86 (±8.53)	12.11 (±7.42)	5.40 (±2.80)	17.92 (±9.18)
	REF2 01	2.10 (±1.38)	3.38 (±0.77)	1.05 (±0.34)	4.41 (±1.73)	49.33 (±19.02)	25.05 (±11.58)	9.86 (±8.55)	5.36 (±2.18)	10.41 (±8.95)
	LOW 01	2.05 (±0.83)	3.27 (±0.44)	0.88 (±0.36)	3.70 (±0.39)	54.30 (±10.48)	10.08 (±6.53)	20.75 (±7.27)	9.12 (±2.48)	5.76 (±3.20)
	HIGH 01	1.41 (±1.07)	1.83 (±1.22)	0.55 (±0.79)	12.87 (±10.11)	25.36 (±24.51)	53.25 (±29.38)	16.20(±6.25)	3.00 (±4.52)	2.19 (±3.72)
2000	REF1 00	1.04 (±1.45)	4.27 (±0.63)	0.36 (±0.56)	2.26 (±0.96)	45.87 (±13.73)	12.92 (±7.37)	4.50 (±3.33)	5.72 (±1.93)	30.99 (±19.6)
	LOW 00	1.78 (±0.51)	3.22 (±0.28)	0.64 (±0.11)	2.90 (±0.25)	51.75 (±5.34)	9.64 (±0.66)	22.37 (±2.06)	9.50 (±1.87)	6.75 (±3.92)
	HIGH 00	1.76 (±0.91)	1.92 (±0.89)	−0.26 (±0.86)	4.30 (±3.03)	34.84 (±22.70)	40.95 (±25.60)	20.72 (±7.42)	2.95 (±1.99)	0.54 (±0.63)

location produced less variability between replicate samples. Reference site 2 was sampled in 2001 and 2002 only, and sediments from this location show more variability between replicates than those found at either Reference site 1 or the site of low dredging intensity. An ordination by PCA of sediment particle size data is illustrated in Figure 3.

In terms of particle size distribution, sediments collected from the area of low dredging intensity and the reference locations were more similar to each other than to sediments from the area of high dredging intensity. This was due to the higher percentage of coarse sand from samples collected from the area of high dredging compared with the samples from the area of low intensity and reference locations. This is reflected in the PCA ordination by the separation of the high intensity samples from the low intensity and reference samples (Figure 4). The particle size distributions of samples from within the area of low dredging intensity were also more consistent over time, as depicted by the tighter clustering of samples in the PCA ordination. In contrast, there was a much higher degree of particle size variability between replicate samples collected from the area of high dredging intensity and the reference locations, as represented by the much wider spread of samples from these locations in the PCA ordination. The separation of sediments collected from the area of low dredging intensity and the reference locations is largely on account of the higher silt/clay content of some of the reference samples (Figure 4).

In all, 70% of the total variation is explained by the first two principal components, indicating that the two-dimensional ordination gives an appropriate representation of the similarity between the collected sediments. Table 3 shows the analysis of similarities results (ANOSIM, Clarke, 1993) for particle size data between samples collected from the different treatments over the 3-year period of study. Sediments at all locations were significantly different ($p < 0.05$) from each other in terms of particle size characteristics, apart from the two reference sites in both 2001 and 2002, the site of higher dredging intensity and Reference site 2 in 2001, and the site of high dredging intensity in 2000 and Reference site 2 in 2001. Sediment characteristics differed over time at each location, although taken together these differences were not, generally, found to be statistically significant at $p < 0.05$.

Macrofaunal assemblage structure

In all, 289 taxa were identified from the 75 Hamon grabs collected from the different treatments. Excepting values of abundance in 2002, population densities and numbers of species of macrofaunal invertebrates were significantly lower ($p < 0.05$) in the site exposed to the highest level of dredging intensity compared with the site of lower dredging intensity and reference conditions (Figure 5). In 2002, higher densities of *Pomatoceros lamarcki* (Quatrefages,

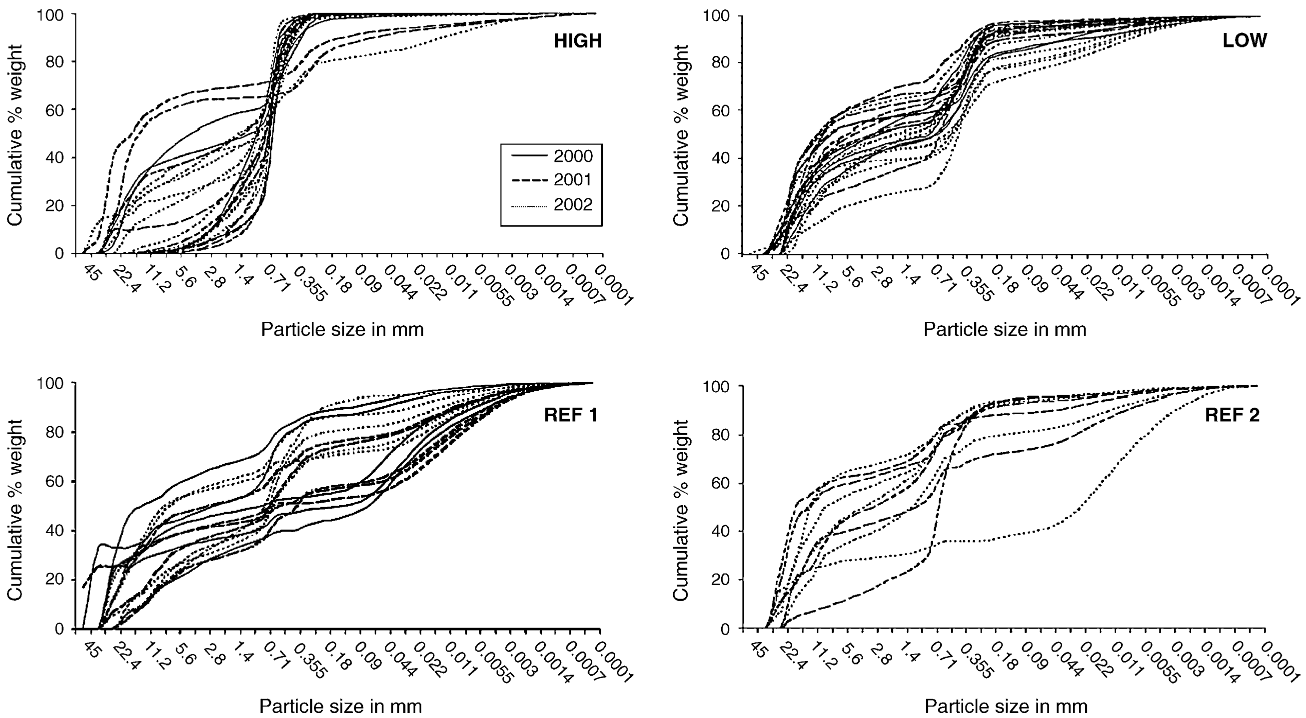

Figure 2. Sediment particle size distributions determined from replicate samples taken from sites of higher and lower levels of dredging intensity and the two references locations.

1866) recorded in one of the samples from Reference site 2 increased the variability around the mean. In this year, therefore, there was no recorded difference between the dredged sites and reference conditions in terms of population densities, although there was still a difference between sites of higher and lower dredging intensity.

In general, differences between the site of higher dredging intensity and other sample locations were due to the absence or reduced abundance ($p < 0.05$) of a range of macrofaunal species characterizing nearby sediments, including the tube worm *P. lamarki*, the pea crab *Pisidia longicornis* (Linnaeus, 1767), the polychaete *Lumbrineris gracilis* (Ehlers, 1868), and the brittle star *Amphipholis squamata* (Chiaje, 1829). Densities of these species were variable between different locations and between different years (Figure 6). Densities of *Amphipholis squamata* increased between 2000 and 2002, while densities of *Lanice conchilega* (Pallas, 1766), a sand-dwelling polychaete were significantly higher ($p < 0.05$) at all sampled sites in 2001.

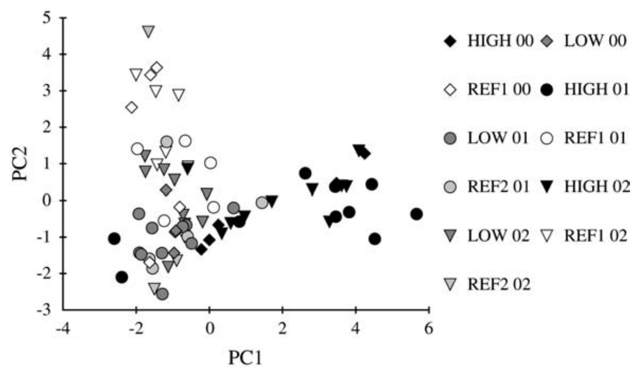

Figure 3. Two-dimensional correlation-based PCA ordination of sediment particle size data from Area 222. Total variance explained by the first two principal components = 70%. For variables involved in the ordination see Table 2.

The MDS ordination for macrofaunal assemblages collected at sites of high and lower dredging intensity and at the two reference sites is presented in Figure 7. While the reference samples and low dredging intensity samples show tight clustering of replicates, indicating a high stability of the spatial pattern, the high intensity replicate samples are much more diffusely distributed. This separation of the individual replicates from the area of high dredging intensity indicates that they are biologically dissimilar.

The comparative Index of Multivariate Dispersion (IMD) has been calculated in order to contrast the multivariate variability among samples taken from the dredged sites with samples from the reference locations (Table 4). Comparisons between the site of high dredging intensity and the reference sites and the sites of high and lower dredging intensity give the most extreme values of IMD, i.e. close to $+1$. In comparison, there is little difference between the low dredging intensity and reference samples in terms of variability in multivariate structure. Thus, a pattern of high variability in multivariate structure with

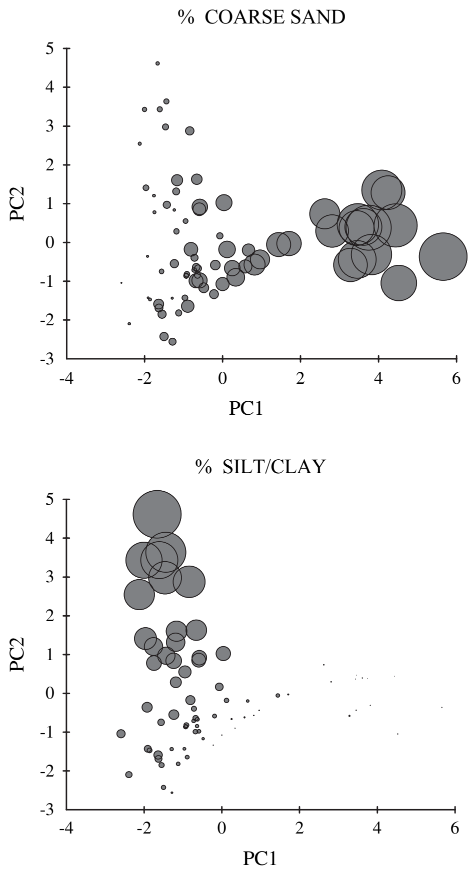

Figure 4. The same two-dimensional correlation-based PCA ordination as in Figure 3, but with superimposed circles proportional in diameter to values of percentage coarse sand and percentage silt/clay.

increased disturbance is clearly evident, in all years, at Area 222. Calculation of the relative Index of Multivariate Dispersion (r.IMD) confirms the conclusions from above, e.g. that there is an increased variability in community composition at the site of high dredging intensity in comparison with the other sampled locations (Table 5).

Macrofaunal assemblages generally discriminated well between different sampling locations in each year. ANOSIM results in Table 6 also confirm the patterns observed in the MDS ordinations. Macrofauna assemblages at all locations were significantly different ($p < 0.05$) from each other in terms of species composition, apart from the two reference sites in both 2001 and 2002, and the site of lower dredging intensity and Reference site 1 in 2001.

Further exploration of the community groupings subject to differing levels of dredging impact was undertaken using the similarity percentages program (SIMPER). Results revealed that the average similarity between replicate samples collected for each of the groups was low, particularly for samples collected from the area of high dredging intensity (see Table 7). This reflects the relatively few shared species found between replicate samples obtained from the area of high dredging intensity.

The output from SIMPER also indicates which taxa contribute the most towards similarity between replicate samples from within each of the groups. Characterizing species from each of the groups were similar over time. From the area of high dredging intensity, characterizing species tended to be infaunal species typically associated with sandy sediments. Juvenile animals also typified high intensity samples. This suggests an active process of recolonization by juvenile animals invading the dredged deposits. In contrast, those species characterizing the areas of low dredging intensity and reference areas were typically larger and included both infaunal and epifaunal species and these species represented a range of different phyla.

Information from SIMPER and ANOSIM also reveal that the differences between the area of high dredging intensity and the reference areas are more pronounced than those between the area of low dredging intensity and the

Table 3. R-values derived from the ANOSIM test for sediment particle size characteristics (mean diameter in mm, sorting coefficient, kurtosis, skewness, % gravel, % coarse sand, % medium sand, % fine sand, and % silt/clay) from locations of higher and lower dredging intensity and from two reference sites in the vicinity of Area 222 sampled in 2000–2002. Performed on normalized Euclidean distance data. Values range between ± 1 and zero. A zero value indicates high similarity, and a value of ± 1 indicates low similarity between samples. *Denotes significant difference at $p < 0.05$ (codes as in Table 1).

	HIGH ’00	LOW ’00	REF1 ’00	HIGH ’01	LOW ’01	REF1 ’01	REF2 ’01	HIGH ’02	LOW ’02	REF1 ’02
HIGH ’00										
LOW ’00	0.480*									
REF1 ’00	0.520*	0.592*								
HIGH ’01	−0.001	0.241*	0.393*							
LOW ’01	0.502*	−0.150	0.650*	0.448*						
REF1 ’01	0.392*	0.544*	0.156	0.393*	0.484*					
REF2 ’01	0.156	0.432*	0.172	0.149	0.401*	0.028				
HIGH ’02	0.004	0.447*	0.674*	0.067	0.614*	0.400*	0.436*			
LOW ’02	0.527*	−0.106	0.681*	0.525*	0.114	0.386*	0.551*	0.605*		
REF1 ’02	0.700*	0.912*	0.012	0.436*	0.808*	0.160	0.476*	0.703*	0.675*	
REF2 ’02	0.268*	0.500*	0.004	0.245*	0.452*	0.128	−0.160	0.521*	0.609*	0.364*

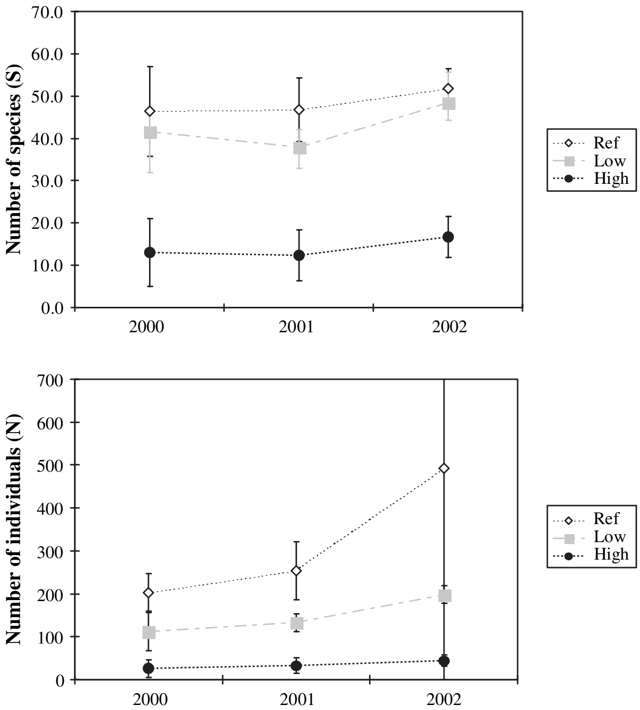

Figure 5. Means (±95% confidence intervals) of numbers of species and numbers of individuals from sites of higher and lower levels of dredging intensity and the nearby reference locations (using Reference site 1 for data arising in 2000, and data for both reference sites combined in 2001 and 2002).

reference areas. These differences in the sample groups are maintained over the 3-year period of study.

Sidescan sonar surveys

Sidescan sonar surveys were conducted at Area 222 in 2000, 2001, and 2002. Figure 8 shows the output of the sidescan sonar survey conducted in 2002. Operational factors such as weather conditions and the acoustic resolution applied which may affect the quality of the acoustic record have been taken into account when comparing the output from the sidescan sonar surveys over time.

The substrata and seabed features within and in the vicinity of Area 222 identified from the 2001 and 2002 sidescan sonar surveys are consistent with those observed in 2000 (Boyd *et al.*, 2003). Disturbed sandy sediments interspersed with patches of sandy gravel and occasional small outcrops of consolidated clay predominate in the northern part of the extraction site. EMS records indicate that this area was subjected to the most intensive dredging activity in the years immediately prior to relinquishment, and some evidence of the effects of the trailer suction hopper dredging consistent with observations made in other studies (e.g. Diesing *et al.*, in press), remains within this part of the site (Figure 9). The sidescan sonar survey conducted in 2001 and 2002 extends to the north of the extraction site and encompasses an area of disturbed seabed previously only surveyed in part in the 2000 survey. This area of seabed to the northeast of the extraction site is uneven, consisting of a series of interconnected pits which is consistent with the effects of static suction hopper dredging (Figure 8). Thus, as noted in previous investigations, it appears that the seabed in this area has been dredged (without a licence) some time prior to the introduction of the EMS in 1993 (Boyd *et al.*, 2003). The area of disturbed seabed extends up to 1000 m away from the northern limit of the extraction site and is characterized by stable slightly muddy sandy gravels, interspersed by patches of clean rippled sand which form the base of the pit structures. Sediment transport features associated with the zone of out of area dredging also appear to extend up to 2500 m away from the northern boundaries of the former extraction site.

A number of large sand waves, whose crests run at right angles to the tidal axis, are present 150 m to the north of Area 222. The presence of these features may be the result of deposition and subsequent entrainment of screened sands produced during the dredging activity within and adjacent to Area 222. Furthermore, the sidescan sonar data indicate that those substrata surrounding Area 222 which have not been directly or indirectly affected by historic dredging activity are similar, being composed of a mixture of sand, gravel, and to a lesser extent silt with the occasional outcrop of clay. It should be noted that while sidescan sonar is effective in describing the nature of the sediment surface, it provides no information on buried substrata.

Changes in the physical status of the seabed have been assessed over the surveyed areas between 2000 and 2002. Particular attention has been given to two areas of seabed that show a persisting physical impact from historic dredging activity (Figures 9 and 10). Sidescan imaging shows little change in the nature and the distribution of the substrata over the wider survey area between 2000 and 2002. In each year, the seabed substrata surrounding the areas of dredging impact (including those found at the reference sites) are stable, mixed sediments.

Within the licensed extraction site, there appears to be some variability in the spatial distribution of sediments over time. Figure 9 shows a sidescan sonar image of the same area of seabed within and immediately surrounding Area 222, in each of the 3 survey years. These images show that while there has been some redistribution of sandy material within the licensed extraction site there has been little change in the overall amount of sand present. Trailer suction hopper dredge tracks are present within and immediately to the north of the extraction site in all years and there appears to be little modification of their appearance over the duration of these surveys (Figures 9 and 10).

Discussion

A number of studies have attempted to identify and explain distribution trends in benthic assemblages following the

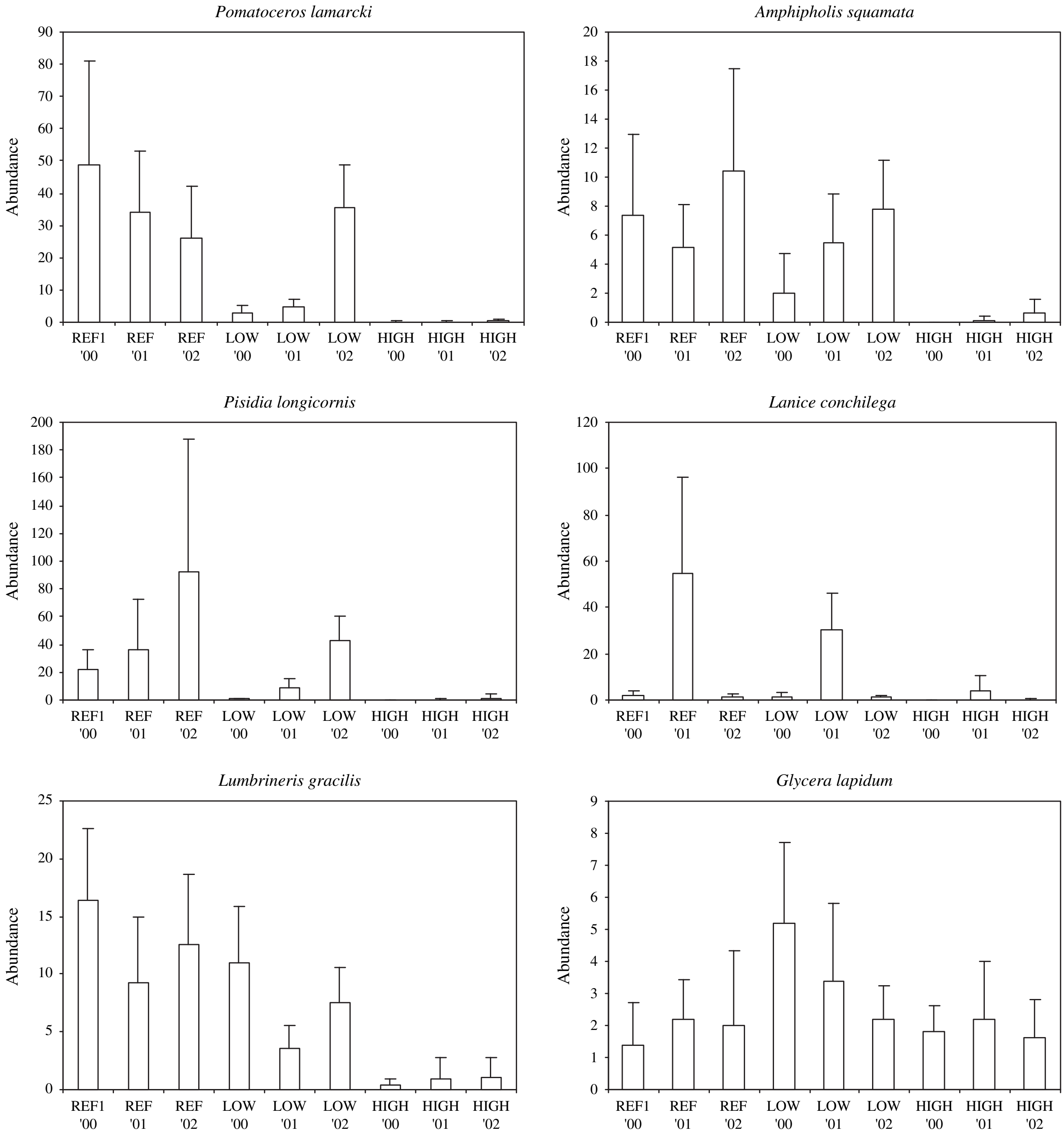

Figure 6. Means (+s.d.) of abundances of selected macrofauna species sampled in 2000–2002 at sites of high and lower levels of dredging intensity and at both reference sites (Codes as in Table 1). Data for 2000 are from Reference site 1, whereas data for 2001 and 2002 are data combined from both reference sites.

cessation of marine dredging (Cressard, 1975; Kenny and Rees, 1994, 1996; Kenny *et al.*, 1998; Desprez, 2000; Sardá *et al.*, 2000; Van Dalfsen *et al.*, 2000; Van Dalfsen and Essink, 2001). Effects on the benthos and sediments identified in such studies show some parallels with the findings observed in this investigation. For example, Desprez (2000) showed that for an industrial extraction site off Dieppe, France, the structure of the benthic community changed from one of coarse sands characterized by the lancelet *Branchiostoma lanceolatum* (Pallas, 1744) to one of fine sands composed of the infaunal polychaetes *Ophelia borealis* (Quatrefages, 1866), *Nephtys cirrosa* (Ehlers, 1868), and *Spiophanes bombyx* (Claparède, 1870). Thus, the change in the assemblage structure reflected a change in sediment composition caused by dredging. Significant changes in particle size composition, resulting in a net fining of the sediment within extraction sites, have also been reported by Van Dalfsen *et al.* (2000) and Sardá *et al.* (2000) following sand extraction.

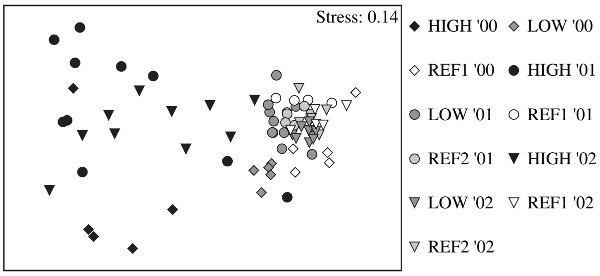

Figure 7. MDS ordination of Bray–Curtis similarities from double square-root transformed species abundance data 4, 5, and 6 years (2000–2002) after the cessation of dredging at high and lower levels of dredging intensity and at two nearby reference locations (codes as in Table 1).

At Area 222, sediments collected from the northern sector of the extraction site tended to contain proportionally more sand and less gravel than most of the other sampled sediments. This part of the extraction site is coincident with the location of intensive dredging recorded by the EMS (see also Boyd *et al.*, 2003). Sidescan sonar images of the extraction site also confirm the presence of dredge tracks within this area. Within the extraction site, screening of the dredged cargoes was undertaken to increase the gravel content with the return of finer material (usually sands) to the sea by means of a reject chute. Over time, this screening activity has the potential to significantly change the composition of sediments within a dredged area. Therefore, it is likely that, over a 25-year period of dredging, the intensively dredged areas of seabed at Area 222 have also undergone a similar transformation to those documented in the literature and thus have become sandier over time.

Table 4. Index of Multivariate Dispersion (IMD) between all pairs of conditions.

Year	Conditions compared	IMD
2000	High/Reference site 1	+0.94
	High/Low	+1
	Low/Reference site 1	–0.04
2001	High/Reference site 1	+0.90
	High/Low	+0.82
	Low/Reference site 1	+0.13
	High/Reference site 2	+0.96
	Low/Reference site 2	+0.51
2002	High/Reference site 1	+0.92
	High/Low	+0.97
	Low/Reference site 1	–0.42
	High/Reference site 2	+0.88
	Low/Reference site 2	+0.63

Table 5. Relative Index of Multivariate Dispersion (r.IMD) in each year.

Year	Site	r.IMD
2000	High intensity	1.558
	Low intensity	0.708
	Reference site 1	0.835
2001	High intensity	1.572
	Low intensity	0.815
	Reference site 1	0.759
	Reference site 2	0.462
2002	High intensity	1.519
	Low intensity	0.372
	Reference site 1	0.653
	Reference site 2	0.771

There is a large variability in the sediment characteristics sampled within the northern part of the extraction site. Presumably, this represents the uneven impact of the dredger draghead on the seafloor. Further evidence of the patchy nature of substrata is provided by sidescan sonar images of the dredged locations. This variability among replicate samples was also evident in biological samples collected from the area of high dredging intensity. Indeed, a high variability in the composition of sediments and benthic assemblages at dredged locations has been reported by Kenny and Rees (1994) and Sardá *et al.* (2000). Such observations lend further support to the hypothesis of Warwick and Clarke (1993), namely that variability in assemblage structure may be an identifiable symptom of perturbed conditions. In this case, however, a higher variability in assemblage structure at the extraction areas appears to be the result of increased habitat heterogeneity within the intensively dredged site which affects both the identity of species and the variation in abundances of each species. This propensity for extraction sites to exhibit variability in terms of sediment characteristics and species composition also has to be referenced against a high degree of natural variability and small-scale sediment patchiness that can be encountered in benthic ecosystems, even at locations like Reference site 2, which appear superficially to be relatively homogeneous.

In contrast to other studies that have demonstrated the rapid degradation of dredge tracks after cessation of dredging (Millner *et al.*, 1977; Kenny *et al.*, 1998), it appears that substantially longer periods, i.e. at least 9 years, are required for the complete erosion of dredge tracks in the disturbed area to the northeast of Area 222. Furthermore, the maintenance of a biological assemblage composed of juvenile animals at the site of high dredging intensity up to 6 years after cessation suggests that these species are unable to reach maturity owing to the unstable nature of sediments in the area. Thus, it appears that at the

Table 6. R-values derived from the ANOSIM test for macrofaunal assemblages (fourth root transformed) from locations of higher and lower dredging intensity and from two reference sites in the vicinity of Area 222 sampled in 2000−2002. Values range between 1 and zero. A zero value indicates high similarity, and a value of 1 indicates low similarity between samples. *Denotes significant difference at $p < 0.05$ (codes as in Table 1).

	HIGH '00	LOW '00	REF1 '00	HIGH '01	LOW '01	REF1 '01	REF2 '01	HIGH '02	LOW '02	REF1 '02
HIGH '00										
LOW '00	0.832*									
REF1 '00	0.944*	0.868*								
HIGH '01	0.297*	0.591*	0.744*							
LOW '01	0.985*	0.748*	0.744*	0.707*						
REF1 '01	0.980*	1.000*	0.824*	0.612*	0.345*					
REF2 '01	0.972*	1.000*	0.808*	0.609*	0.162	0.188				
HIGH '02	0.508*	0.480*	0.751*	0.190*	0.665*	0.596*	0.601*			
LOW '02	0.993*	0.995*	0.909*	0.797*	0.612*	0.889*	0.785*	0.742*		
REF1 '02	0.980*	1.000*	0.668*	0.754*	0.717*	0.536*	0.652*	0.614*	0.560*	
REF2 '02	0.980*	0.972*	0.720*	0.345*	0.615*	0.548*	0.528*	0.560*	0.593*	0.136

site of high dredging intensity the effects of dredging are still discernible on the composition of sediments and fauna even 6 years after cessation. This is in direct contrast to a body of case studies which together suggest that substantial progress towards restoration of the fauna could be expected within 2−4 years following cessation of marine sand and gravel extraction (Millner *et al.*, 1977; Kenny *et al.*, 1998; Desprez, 2000; Sardá *et al.*, 2000; Van Dalfsen *et al.*, 2000; ICES, 2001). This discrepancy between the Area 222 data and other studies may reflect differences in the magnitude of dredging disturbance, since many of the studies reported in the literature have been concerned with the effects of relatively short-lived dredging campaigns (Kenny *et al.*, 1998; Sardá *et al.*, 2000; Van Dalfsen *et al.*, 2000), whereas Area 222 was dredged repeatedly over a 25-year period. Indeed, evidence from the current study suggests that the nature and speed of recolonization is affected by the intensity of dredging. This may also suggest that undisturbed deposits between dredged furrows provide an important source of colonizing species (Newell *et al.*, 1998), allowing recolonization to proceed more rapidly in less heavily dredged sediments than in areas of intensive dredging. The discrepancy between our findings and those reported in the literature is also likely to reflect differences in the sediment composition at the dredging sites. Sites with sediments containing a higher gravel content typically support a richer assemblage than sandy substrata, and therefore it is to be anticipated that such sites will require a longer time scale for successful regeneration of benthic assemblages (see later).

Local environmental factors are also likely to affect the rate of re-establishment of benthic fauna and in particular the hydrodynamics of a site determine the sedimentary characteristics of an area and will ultimately be responsible for determining broad-scale community patterns (Warwick and Uncles, 1980; Hall, 1994; Rees *et al.*, 1999). It is therefore apparent that any changes in the status of benthic assemblages in areas which have been subjected to extraction will need to be referenced against both variations in particle size and the hydrodynamic regime. Based on the water depth, tidal current, and wave data (Admiralty data), Area 222 is considered to be a site of "moderate" energy. This conclusion is important since it implies that many years, possibly decades, will be required for re-establishment of benthic assemblages in sites classed as "low energy".

The recolonization of dredged sites

From studies of dredged sites (Cressard, 1975; Kenny and Rees, 1994, 1996; Kenny *et al.*, 1998; Desprez, 2000; Sardá *et al.*, 2000; Van Dalfsen *et al.*, 2000; Van Dalfsen and Essink, 2001; Boyd *et al.*, 2003) and from observations following defaunation as a consequence of storm disturbance (Rees *et al.*, 1977), a general pattern of recolonization is emerging (see also ICES, 2001). The first stage involves the settlement of a few opportunistic species, which are able to take advantage of the dredged and sometimes unstable sediments (Hily, 1983; Kenny and Rees, 1994, 1996; Kenny *et al.*, 1998; Desprez, 2000; Van Dalfsen *et al.*, 2000; Van Dalfsen and Essink, 2001). Recolonization can either be by adults or larvae from the surrounding area if the sediments of the disturbed area are similar to the original substrata (Cressard, 1975) or by larvae from more distant sources if the sediment is markedly different (Santos and Simon, 1980; Hily, 1983). These species can substantially increase the overall abundance and the numbers of species during the early stages of post-dredging recolonization (Hily, 1983; López-Jamar and Mejuto, 1988; Kenny and Rees, 1994, 1996; Kenny *et al.*, 1998; Desprez, 2000; Van Dalfsen *et al.*,

Table 7. Results from SIMPER analysis of macrofauna data from Area 222 (all taxa excluding colonial species, fourth root transformed), listing the main characterizing species from samples subject to differing levels of dredging impact from 2000 to 2002. Average abundance, average similarity and the % contribution to the similarity made by each characterizing species is shown. Also listed is the cumulative percentage and the overall average similarity between replicate samples from within each group.

Group	Taxonomic group	Average abundance	Average similarity	% Contribution	Cumulative %	Overall average similarity %
HIGH ’00	*Glycera lapidum* (agg.)	1.8	7.98	26.90	26.90	29.7
	Sphaerosyllis taylori	2.4	4.29	14.47	41.37	
LOW ’00	*Lumbrinereis gracilis*	11.0	3.45	6.54	6.54	52.73
	Exogone verugera	5.4	2.80	5.30	11.84	
	Glycera lapidum (agg.)	5.2	2.79	5.28	17.13	
	Notomastus sp.	4.8	2.65	5.02	22.15	
	Ophiuroidea (juv.)	3.6	2.64	5.01	27.16	
	Polycirrus spp.	4.6	2.51	4.76	31.92	
	Spiophanes bombyx	2.8	2.28	4.32	36.24	
	Ascidiidae (juv.)	2.6	2.26	4.29	40.53	
REF1 ’00	*Pomatoceros lamarcki*	48.6	3.81	7.66	7.66	49.70
	Pisidia longicornis	22.2	3.30	6.64	14.30	
	Lumbrinereis gracilis	16.4	3.28	6.60	20.90	
	Polydora sp.	9.8	2.84	5.71	26.61	
	Amphipholis squamata	7.4	2.35	4.72	31.33	
	Sabellaria spinulosa	5.2	2.24	4.51	35.84	
	Harmothoe spp.	6.8	2.04	4.10	39.94	
	Sphaerosyllis taylori	2.8	1.94	3.90	43.85	
HIGH ’01	*Spisula* sp. (juv.)	3.9	8.49	31.21	31.21	27.20
	Glycera lapidum (agg.)	2.2	8.39	30.82	62.03	
LOW ’01	*Lanice conchilega*	30.4	4.38	8.83	8.83	49.65
	Pisidia longicornis	8.9	3.08	6.20	15.03	
	Amphipholis squamata	5.5	2.98	6.01	21.04	
	Harmothoe spp.	4.4	2.74	5.53	26.56	
	Echinocymanus pusillus	4.9	2.71	5.47	32.03	
	Lumbrineris gracilis	3.6	2.68	5.40	37.43	
	Pomatoceros lamarcki	4.6	2.26	4.55	41.98	
REF1 ’01	*Pisidia longicornis*	55.0	4.21	8.24	8.24	51.13
	Lanice conchilega	25.4	3.61	7.07	15.31	
	Pomatoceros lamarcki	30.2	3.55	6.95	22.26	
	Serpulidae	14.80	3.36	6.57	28.83	
	Amphipholis squamata	6.0	2.64	5.17	34.00	
	Gibbula sp.	3.6	2.30	4.49	38.49	
	Harmothoe spp.	5.6	2.24	4.38	42.86	
REF2 ’01	*Lanice conchilega*	83.6	4.26	7.61	7.61	56.03
	Pomatoceros lamarcki	38.4	3.52	6.29	13.90	
	Serpulidae	12.6	2.73	4.87	18.77	
	Lumbrineris gracilis	10.8	2.53	4.52	23.28	
	Pisidia longicornis	18.6	2.44	4.35	27.63	
	Scalibregma inflatum	9.0	2.44	4.35	31.98	
	Lagis koreni	5.6	2.10	3.75	35.73	
	Cerianthus lloydi	4.4	2.02	3.60	39.34	
	Gibbula sp.	6.0	2.01	3.59	42.93	
HIGH ’02	*Spisula* sp. (juv.)	6.6	7.18	23.68	23.68	30.30
	Nemertea	2.8	4.74	15.64	39.33	
	Glycera lapidum (agg.)	1.6	3.27	10.80	50.13	
LOW ’02	*Pisidia longicornis*	43.0	3.98	6.89	6.89	57.76
	Pomatoceros lamarcki	35.5	3.78	6.54	13.43	
	Sepulidae	8.7	2.62	4.54	17.97	

Table 7 (*continued*)

Group	Taxonomic group	Average abundance	Average similarity	% Contribution	Cumulative %	Overall average similarity %
	Amphipholis squamata	7.8	2.60	4.51	22.48	
	Lumbrineris gracilis	7.5	2.52	4.37	26.85	
	Scalibregma inflatum	7.3	2.39	4.13	30.99	
	Caulleriella alata	7.6	2.38	4.11	35.10	
	Notomastus	6.3	2.14	3.70	38.81	
	Echinocyamus pusillus	4.5	2.09	3.63	42.43	
REF1 '02	*Pisidia longicornis*	155.4	4.58	8.73	8.73	52.45
	Pomatoceros lamarcki	25.0	2.82	5.37	14.10	
	Amphipholis squamata	15.0	2.75	5.24	19.34	
	Lumbrineris gracilis	12.8	2.46	4.69	24.03	
	Serpulidae	6.0	2.18	4.16	28.19	
	Cheirocratus sp.	3.8	2.02	3.85	32.04	
	Laonice bahusiensis	4.2	1.96	3.73	35.78	
	Corophium sextonae	4.0	1.81	3.44	39.22	
	Caulleriella alata	5.6	1.73	3.31	42.53	
REF2 '02	*Pomatoceros lamarcki*	27.0	3.35	6.62	6.62	50.61
	Lumbrineris gracilis	12.4	2.91	5.76	12.38	
	Pisidia longicornis	28.4	2.72	5.37	17.75	
	Serpulidae	9.6	2.69	5.31	23.06	
	Ampelisca spinipes	7.2	2.54	5.01	28.07	
	Laonice bahusiensis	3.6	2.05	4.05	32.12	
	Galathea intermedia	3.4	2.02	3.99	36.11	
	Amphipholis squamata	5.8	1.95	3.86	39.97	
	Scalibregma inflatum	3.8	1.92	3.80	43.77	

2000). A second phase is characterized by a reduced community biomass which can persist for a number of years (Kenny and Rees, 1994, 1996; Kenny *et al.*, 1998; Desprez, 2000). There is a natural expectation that biomass will remain reduced, while new colonizers "grow on" to maturity comparable with the pre-dredging age/size profile (Rees, 1987; Van Dalfsen and Essink, 2001). Furthermore, a reduced biomass may also be caused by the abrasive effects of increased sediment (mainly sand) inhibiting the growth and survivorship of epifauna such as hydroids and bryozoans. Paradoxically, it is this sandy sediment that is also responsible for the infilling of dredge tracks (Kenny *et al.*, 1998; ICES, 2001; Limpenny *et al.*, 2002), which in the longer term may promote physical stability. Over time, it may be expected that, at some sites, the bedload transport will approach the pre-dredged equilibrium, allowing the restoration of community biomass (Kenny *et al.*, 1998). A similar model of response has been represented schematically by Hily (1983) and includes a further stage in which opportunists are replaced by a greater number of species. It was suggested that this replacement was the result of increasing levels of interspecific competition. However, this model was based on observations following the dredging of a sandy mud (Hily, 1983), and further evidence is required to establish whether such oscillations occur in more stable gravel habitats during the latter stages of succession. Evidence from Area 222 suggests that, where there are significant changes to the topography and composition of the sediments as a result of dredging activity, the maintenance of a biological community, over prolonged periods, at an early developmental stage can be expected.

Framework for future studies

As yet, coordinated studies on a wide geographical scale investigating the physical and biological status of commercial aggregate extraction sites in the UK and elsewhere are limited (Van Dalfsen *et al.*, 2000). One consequence of the limited available information on the effects on the benthos of marine aggregate extraction is the difficulty it creates for the establishment of reliable empirical models for predictive purposes. A further difficulty in generalizing about the effects of extraction is the variability in both the dredging history and the particular dredging practices to which different extraction sites are exposed, i.e. a typology of dredging disturbance does not exist. Consequently, when seeking to develop and then apply predictive models, generalizations about the effects of marine aggregate extraction must be qualified by local information regarding the nature of dredging activity and the conditions under which extraction activity occurs. Based on existing evidence, however, the two most

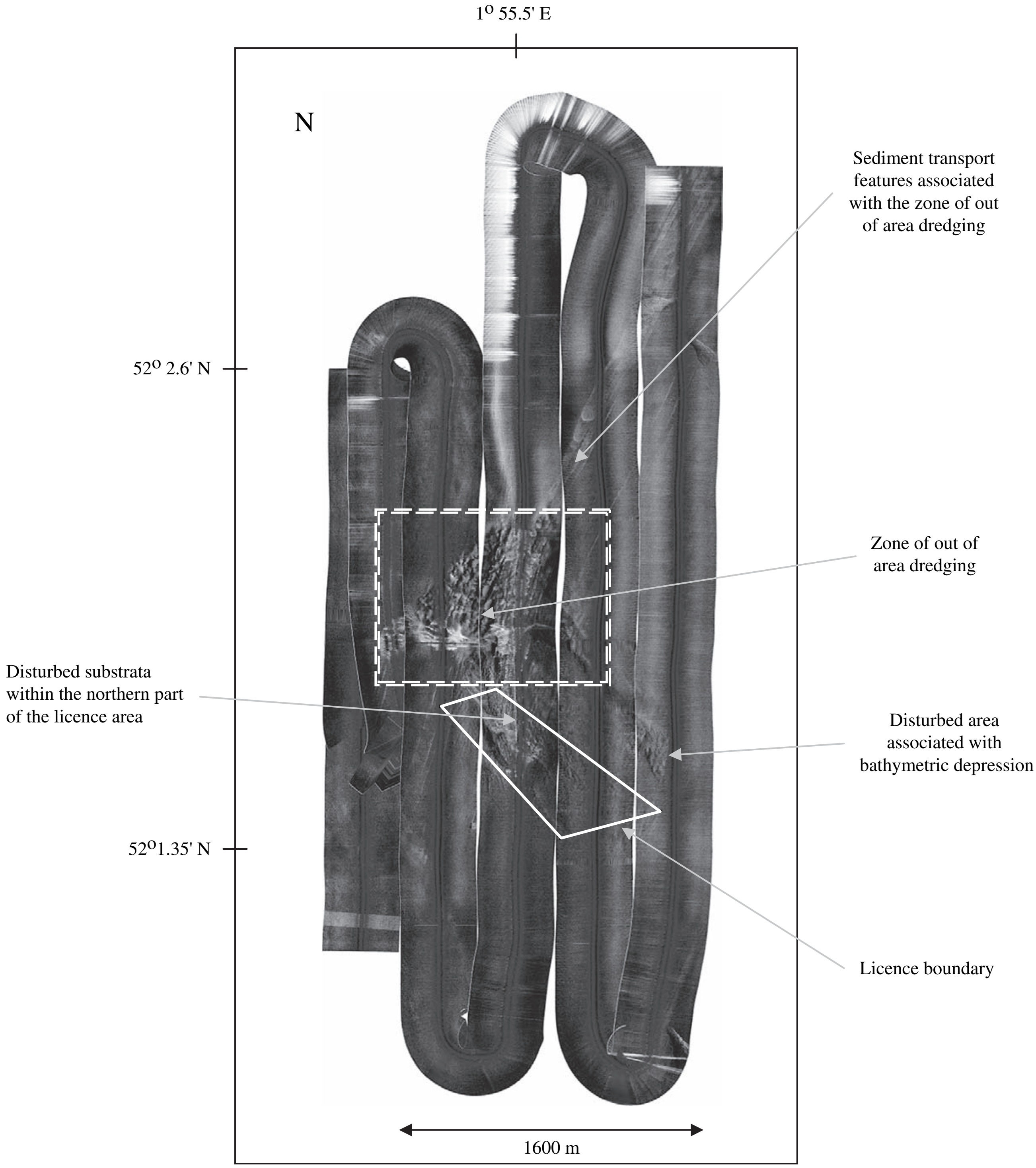

Figure 8. A sidescan sonar mosaic derived from a survey undertaken in 2002 showing the distribution of substrata within and surrounding the former extraction site at Area 222, southern North Sea.

commonly encountered scenarios following marine aggregate extraction in the UK are:

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

This is not to exclude the possibility of other consequences such as the exposure of clay depending on local circumstances (Boyd *et al.*, 2003). In the first of the scenarios, there are a number of ways in which alterations to the sediment as a consequence of dredging could result. These include, but may not be limited to, the exposure of an underlayer of finer

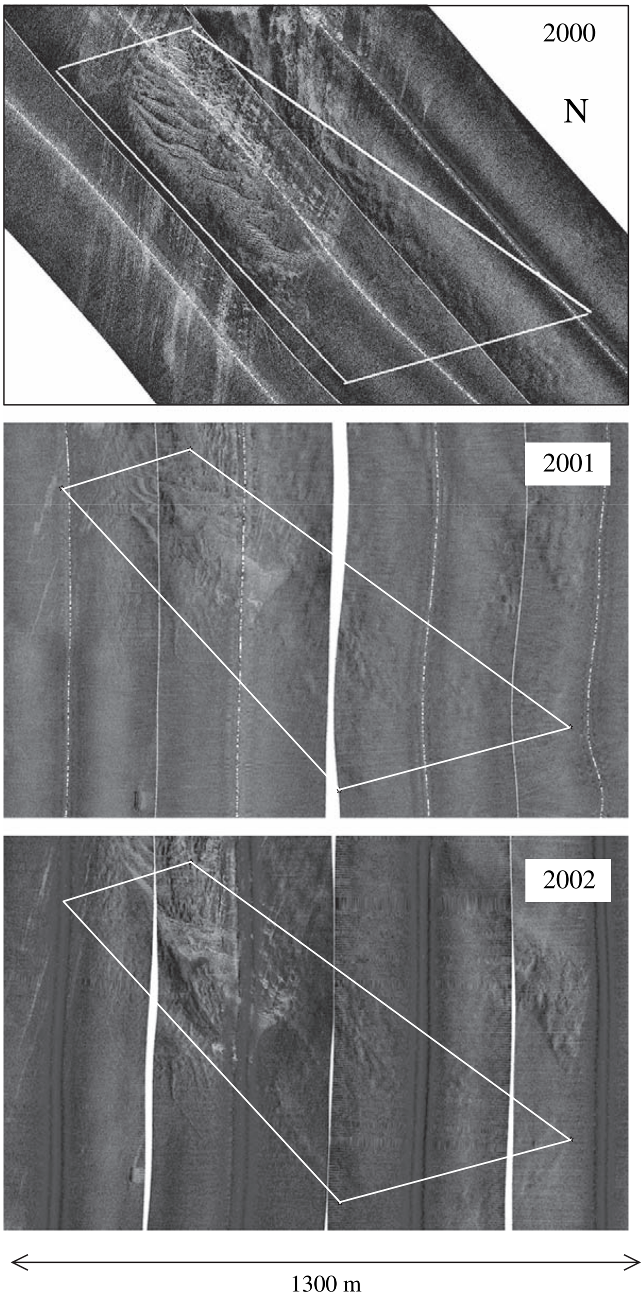

Figure 9. Sidescan sonar mosaics of the relinquished extraction site, collected in 2000, 2001, and 2002. The boundary of the former extraction licence is outlined in white.

sediments, discharge of finer sediments from spillways (Hitchcock and Drucker, 1996; Van Dalfsen *et al.*, 2000) or screening and the trapping of bedload in dredged furrows (Desprez, 2000; Sardá *et al.*, 2000; ICES, 2001). The degree of change appears to depend both on the local circumstances (Desprez, 2000; Van Dalfsen *et al.*, 2000; Boyd and Rees, 2003), and on the magnitude of perturbation i.e. differences in the intensity, type of dredging, or length of extraction period (Van Dalfsen *et al.*, 2000; Boyd *et al.*, 2003; Boyd and Rees, 2003). The colonizing fauna also appear to reflect this change to the substrata, through a shift in the proportions of sandy vs. gravelly fauna (Desprez, 2000). Accompanying this, it is postulated that there would be a net decline in biomass. This model of response is portrayed schematically in Figure 11. A similar model of response could account for changes at some sand extraction sites where the seabed substrata have changed from coarse to fine sands (Sardá *et al.*, 2000; Van Dalfsen *et al.*, 2000).

In the second scenario, sediments present at the seabed following the cessation of marine aggregate extraction are similar to those which existed prior to disturbance, i.e. sandy gravels. This scenario accords with current expectations regarding seabed status following licensed dredging and is consistent with the management aim of ensuring that the seabed environment is left in a comparable physical condition to that prevailing prior to the onset of dredging i.e. with a similar sediment type and evenness profile. From the limited available data concerning the effects of marine gravel extraction (Kenny and Rees, 1994, 1996; Kenny *et al.*, 1998), it is reasonable to postulate that the fauna recolonizing such sites will follow classical successional dynamics (e.g. Grassle and Sanders, 1973). Indeed, this scenario was postulated in connection with the disposal of dredged coarse material arising from a port expansion or channel deepening (see Anon., 1996). Although such simplified models require further validation and/or refinement, they provide a useful framework for evaluating the outcome of post-cessation recolonization studies and recovery rates and eventually could provide a reliable predictive capability. A body of case studies on the consequences of marine aggregate extraction over sufficiently long time scales from sites exposed to commercial extraction practices is therefore required to underpin the derivation of reliable and scientifically credible models of response. Such a need applies equally to many other human activities which take place in the marine environment.

Conclusions

Many of the field studies reported in the literature are the results of investigations on the impacts of short-term dredging events, and these have proved useful in determining the rates and processes leading to benthic re-establishment following aggregate extraction. From such studies and those undertaken at sites exploited for commercial interests, a general pattern of response to marine aggregate extraction is emerging. This needs to be tested to establish its general validity in all environments, particularly in areas which have been exposed to industrial scale dredging operations over many years. From such work, it is clear that re-establishment of a community similar to that which existed prior to dredging can only be attained if the topography and original sediment composition are restored (for a contrary view, see Seiderer and Newell, 1999). Should physical stability of the sediments not be attained, however, it is hypothesized that communities will remain at an early developmental stage.

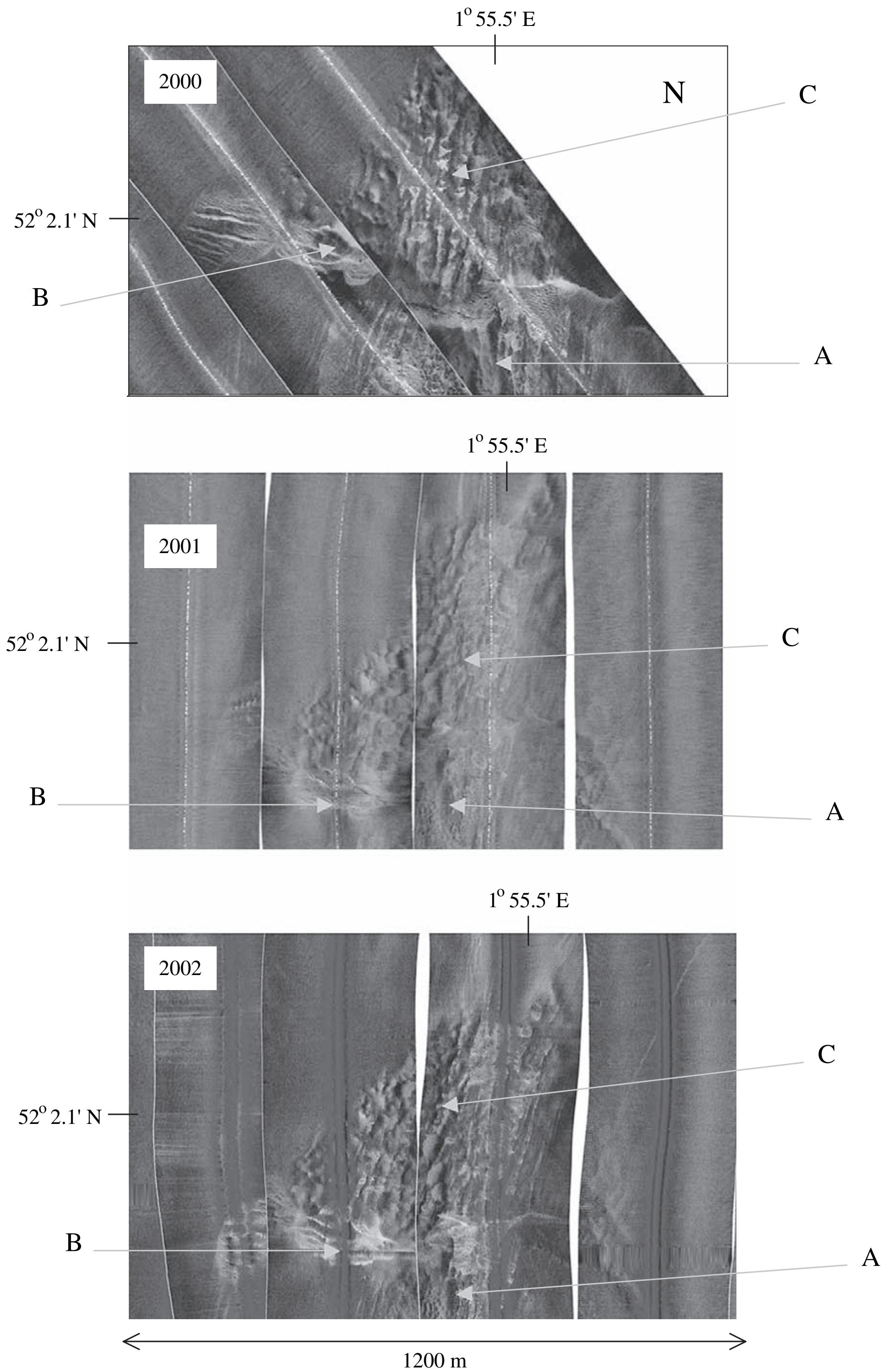

Figure 10. Sidescan sonar images collected between 2000 and 2002 of the disturbed area of seafloor to the north of Area 222, A) evidence of suction trailer dredging, B) large sand waves, C) evidence of static suction dredging.

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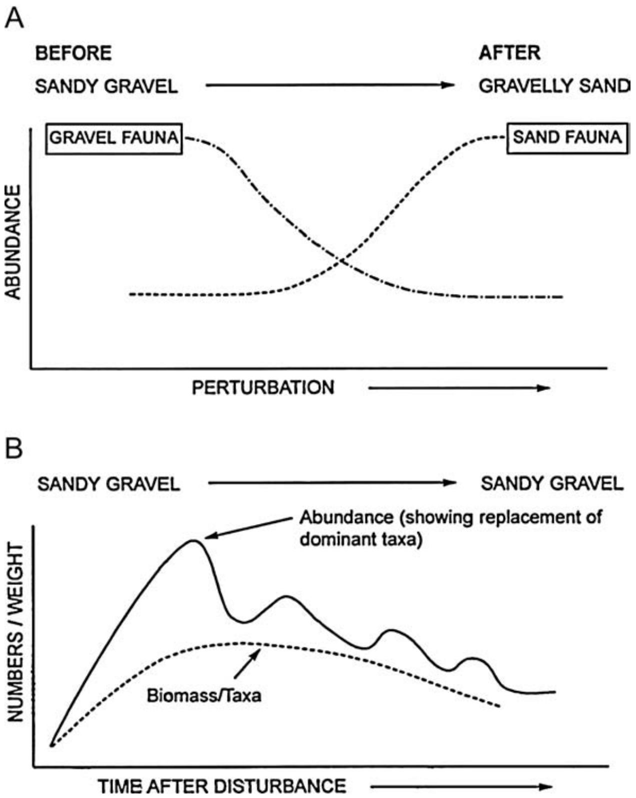

Figure 11. A) Simplified diagram of changes in the proportions of gravelly fauna in response to a change in sediment type as a consequence of marine aggregate extraction. B) Simplified model of changes in the benthos after the cessation of marine aggregate extraction.

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Paper #13

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Mapping seabed biotopes at Hastings Shingle Bank, eastern English Channel. Part 1. Assessment using sidescan sonar

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A multi-technique approach was used to map the spatial distribution of seabed biotopes (i.e. physical habitats and their associated benthic assemblages) in the vicinity of Hastings Shingle Bank in the eastern English Channel, part of which is licensed for the extraction of marine aggregates for the construction industry. An area of seabed, approximately 12×4 km in size, was surveyed using a high-resolution sidescan sonar system, and a mosaic of the output was produced, covering 100% of the survey area. The area was then divided into acoustically distinct regions based on the sidescan sonar data, and the benthic communities and sediment types within each of the regions were ground-truthed using a Hamon grab fitted with a video camera, and using a heavy duty 2-m beam trawl. Additional information concerning the seabed was obtained through the application of video and photographic techniques. Sediments within the survey area ranged from cobbles and coarse gravels on the Shingle Bank, to various grades of sands to the north and south. Analysis of faunal data revealed the presence of statistically distinct biological assemblages within each acoustic region. Using all available data, four discrete biotopes were identified and their spatial distribution mapped across the survey area.

INTRODUCTION

There are many sonar devices currently on the market which can be used to map various seabed properties (e.g. sediment type, topography, surface texture). These acoustic systems can generally be divided into the following categories: (a) broad-acoustic beam (swath) systems such as sidescan sonar; (b) single beam acoustic ground discrimination systems (AGDS) such as RoxAnn and QTC-View; (c) multiple beam swath bathymetric systems; and (d) multiple beam (interferometric) sidescan sonar systems (Kenny et al., 2000). Recent improvements in many of these acoustic systems in the 1990s, in particular with swath and multibeam systems as a result of increased digital processing power offered by modern computers, have led to very high resolution and affordable systems entering the market place. This development is reflected in the number of recent investigations which have used acoustic techniques as a means to infer the biological status of the seabed (e.g. Magorrian et al. (1995) and Greenstreet et al. (1997) using RoxAnn systems; Wildish & Fader (1998) and Tuck et al. (1998) using sidescan sonar; Kostylev et al. (2001) using multibeam bathymetry). Although the outcomes of these studies are, in general, encouraging, the approaches have not yet reached the stage of uncritical, routine application. However, these developments are offering the opportunity for researchers to move away from a process of inference around a matrix of spot samples into the realm of spatially continuous mapping using spot sampling for ground-truthing. For this reason the use of acoustic techniques to assist in mapping the geographical distribution of biotopes (e.g. physical habitats and associated biological communities) can be seen to have many potential advantages, including the prospect of 100% coverage of the seabed as resources allow or priorities dictate.

The choice as to which acoustic system should be used depends on a number of factors: (1) which properties of the seabed are to be measured (e.g. bathymetry, surface texture, sediment type); (2) the area of seabed to be covered; (3) whether or not 100% coverage is required from the system; (4) the cost of the system. Whilst many of the acoustic techniques have been proven to effectively map the surface geology of the seabed, the extent to which they can be used for mapping the spatial distribution of biotopes is still unclear.

The work described in this paper formed part of a wider study, funded by the UK Department for Environment, Food and Rural Affairs (Defra), which aimed to evaluate the utility of a number of acoustic systems for mapping seabed biotopes in areas of coarse substrates. In this paper, high-resolution sidescan sonar is used to map seabed biotopes at relatively fine scales at a site in the eastern English Channel adopting an integrated approach similar to that described in Brown et al. (2002), and the results describe the spatial distribution of biotopes within this region.

MATERIALS AND METHODS

Sidescan sonar survey

The survey site crossed Hastings Shingle Bank in the eastern English Channel, covering an area of approximately 12×4 km (Figure 1). A sidescan sonar

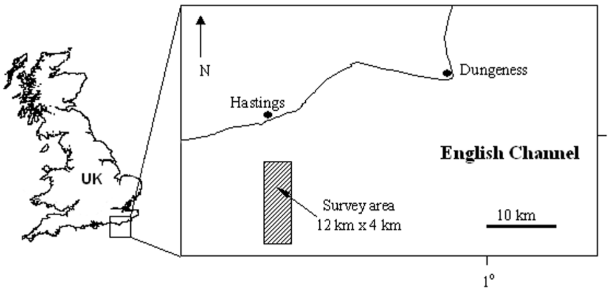

Figure 1. Geographical location of the survey area.

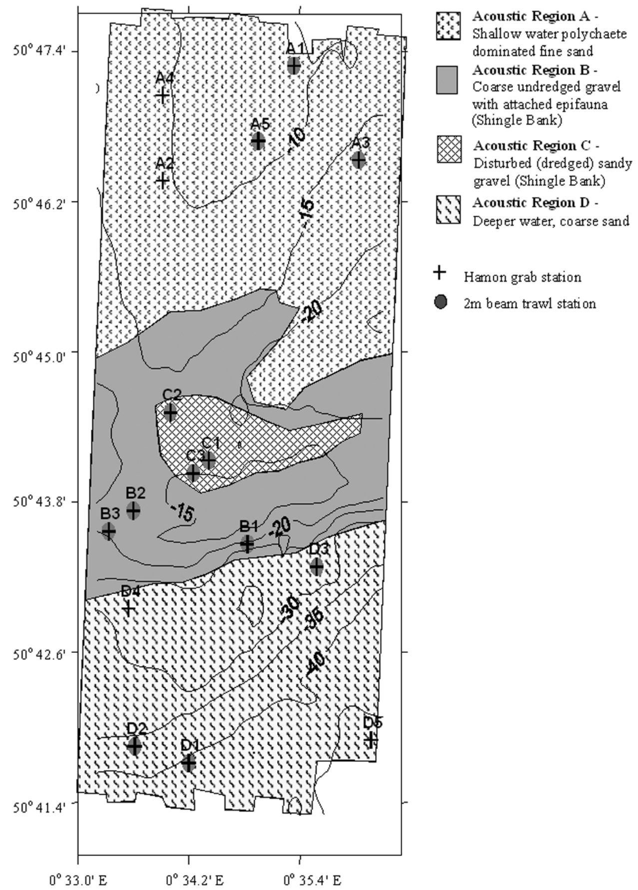

Figure 2. Plot of the survey area showing the four acoustically distinct regions (A, B, C and D) determined from the sidescan sonar data, and locations of the sampling stations.

survey of the site was carried out in July 1999 using a Datasonics SIS1500 digital chirps sidescan sonar with a Triton Isis logging system. Delphmap post-processing software was used to mosaic the imagery and classify texturally different regions. The system was operated on a 400 m swath range, and survey lines were spaced at 400 m intervals in a north–south orientation in order to ensonify 100% of the survey area. Vessel position was provided by the Veripos differential global positioning system (DGPS) and towed sensor position calculated by vessel heading, towcable layback and towfish depth, all of which were logged in real time by the Isis system. A drop-camera frame fitted with an under-water video camera and light was deployed at 12 stations across the survey area in order to provide visual ground-truth data to aid interpretation of the sidescan sonar data set.

Seabed features (rippled substrata, rough uneven topography, dredged tracks etc.) and an indication of the sediment type (soft or hard sediments) could be identified from the sidescan sonar backscatter, and the presence of these features/characteristics was confirmed through the underwater video data collected at the ground-truth stations. The survey area was divided into four acoustically distinct regions based on information derived from the sidescan mosaic and the underwater video data. These regions formed the basis for the design of subsequent biological and sedimentological surveys.

Benthic survey

The design of the biological and ground-truthing survey was structured around the four acoustically distinct regions identified from the output of the sidescan sonar survey. The main sampling tool was a 0.1 m^2 Hamon grab fitted with a video camera and light. This was the preferred type of sampling gear due to its ability to collect samples on coarse, unconsolidated sediments. The grab was fitted with a video camera in order to record an image of the seabed adjacent to the collection bucket of the grab, thus providing information about the undisturbed surface of the substrate at each sampling station. Sampling stations were randomly positioned within each of the four acoustic regions, and the number of stations within each region was linked to the size of the area (Figure 2).

A total of 16 Hamon grab samples was collected from across the study area in October 1999. Following estimation of the total volume of each grab sample, a 500 ml sub-sample was removed for laboratory particle size analysis. The remaining sample was washed over 5 mm and 1 mm square mesh sieves to remove excess sediment. The retained macrofauna were fixed in 4–6% formaldehyde solution (diluted with seawater) for laboratory identification and enumeration.

A 2 m beam trawl survey was also conducted in order to characterize the epifauna (July/August 2000). A modified 2 m beam trawl, with a heavy-duty steel beam, chain mat and a 4 mm knotless mesh liner fitted inside the net was deployed at selected sampling stations within each of the acoustically distinct regions. The beam trawl was deployed from the stern ramp of the research vessel using a warp length of three times the water depth. Each tow covered a fixed distance of 120 m across the seabed, which was determined using Sextant software linked to the ship's DGPS. The speed of the ship and the deployment time were also recorded. On retrieval of the trawl each sample was washed over a 5-mm square mesh sieve and macrofaunal species were identified and enumerated at sea. Colonial species were recorded as either present or absent. Any specimens that could not be identified at sea were fixed in formaldehyde solution and returned to the laboratory for identification.

A drop-camera frame fitted with a video camera and lights was deployed at a number of stations to obtain additional, visual, qualitative ground-truth data from each of the acoustic regions. The camera system was suspended above the surface of the seabed (no greater than 2 m from the seabed) as the vessel was allowed to drift. Deployments were made around slack water when current speeds were at their lowest in order to achieve good quality video footage.

In the laboratory, Hamon grab samples were first washed with freshwater over a 1-mm square mesh sieve in a fume cupboard to remove the excess formaldehyde solution. Samples were then sorted and the specimens placed in jars or Petri dishes containing a preservative mixture of 70% methanol, 10% glycerol and 20% tap-water. Specimens were identified to species level, as far as possible, using standard taxonomic keys. The number of individuals of each species was recorded, and colonial species were recorded as present or absent. For each positive identification a representative specimen was retained in order to establish a reference collection.

The sediment sub-samples from each grab station were analysed for their particle size distributions. Samples were first wet sieved on a 500 micron stainless steel test sieve, using a sieve shaker. The sediment fraction less than 500 microns, along with water from the wet sieving, was allowed to settle in a bucket for 48 hours. Excess water was then removed using a vacuum pump and the fraction was washed into a sterile Petri dish, frozen for 12 h and freeze dried. The weight of the sediment was also recorded. A sub-sample of the <500 micron freeze dried fraction was then analysed on a laser sizer. The >500 micron fraction was washed from the test sieve into a foil tray and oven dried at $\sim 90^{\circ}$C for 24 hours. It was then dry sieved for 10 min on a range of stainless steel test sieves at half phi intervals, down to 1 phi. The sediment on each sieve was weighed to 0.01 g and the results recorded. The results from these analyses were combined to give the full particle size distribution. The mean and sorting values were then calculated.

Data processing

Total number of individuals (excluding colonial species) and total number of species were calculated from both the Hamon grab and beam trawl surveys as summary measures of benthic assemblages within each acoustic region. Associations between benthic assemblages and acoustic regions were examined using multivariate statistical methods. Analysis was conducted on the entire dataset excluding colonial taxa. Sample and species associations across the survey area were assessed by non-metric multi-dimensional scaling (MDS) ordination using the Bray–Curtis similarity measure on 4th root transformed data using the software package PRIMER (Clarke & Warwick, 1994). Rare species (i.e. with fewer than three individuals recorded throughout the survey area) were removed from the analysis in order to reduce the variability caused by these infrequently occurring species. Removing these species was also necessary to conform to certain limitations in the total number of species which can be used during certain tests within the PRIMER software (e.g. SIMPER—see below). The majority of species collected during the beam trawl surveys were epifaunal species. Statistical analysis was therefore conducted on all taxa excluding colonial organisms using identical statistical methods as above on 4th root transformed data.

Analysis of similarities (ANOSIM, Clarke, 1993) was performed to test the significance of differences in macrofauna assemblage composition between samples. The nature of the groupings identified in the MDS ordinations were explored further by applying the similarity percentages program (SIMPER) to determine the contribution of individual species to the average dissimilarity between samples.

A correlation-based principal components analysis (PCA) was applied to ordinate results from the sediment particle size analysis. Prior to analysis, environmental variables were converted to approximate normality using a log $(1+N)$ transformation. Analysis of similarities (ANOSIM, Clarke, 1993) was performed on particle size data to test the significance of differences in particle size composition between acoustic regions.

RESULTS

Acoustic data interpretation

This survey site crossed Hastings Shingle Bank, parts of which have been licensed for some years for the commercial extraction of marine aggregates for the construction industry. The survey therefore had the additional benefit of allowing an evaluation of the success of the techniques in identifying any consequences of man-made perturbations at the seabed. The structure of the bank was clearly discernible from the sidescan mosaic. Examination of these data revealed the presence of four acoustically distinct regions (labelled A, B, C and D) within the survey area (Figure 2). The Shingle Bank could be divided into two regions which, following ground-truthing with the underwater video camera, related to areas of coarse gravel (Region B) and of dense dredge tracks in coarse gravel infilled with sand and silt (Region C). The regions to the north and south of the Shingle Bank both appeared from the sidescan record to consist of rippled sand. However, ground-truthing revealed that the inshore region consisted of fine–medium rippled sand at water depths of less than 20 m (Region A), whereas the offshore region was predominantly slightly gravelly rippled sand at water depths greater than 20 m (Region D). Boundaries between adjacent regions were clearly defined, and the substrata within Regions A, B and D tended to be homogeneous in their distribution. Examples from the sidescan record and images taken from the underwater video footage of each acoustically distinct region are illustrated in Figure 3.

Sediment characteristics and environmental variables

Examination of the grab samples on deck, and *in situ* study of the undisturbed seabed surface by the video camera attached to the side of the grab, confirmed the interpretations from the acoustic data. Results from the particle size analysis of grab samples, used in conjunction with information derived from the sidescan sonar mosaic and video footage, provided a clear understanding of the physical habitat characteristics within each acoustic region.

Samples collected from the Shingle Bank (Regions B and C) had a much higher percentage of coarse material than samples collected from regions to the north and south of the bank (A and D), which consisted mainly of sand. This is reflected in the PCA ordination by the separation of A and D from B and C (Figure 4). The particle size

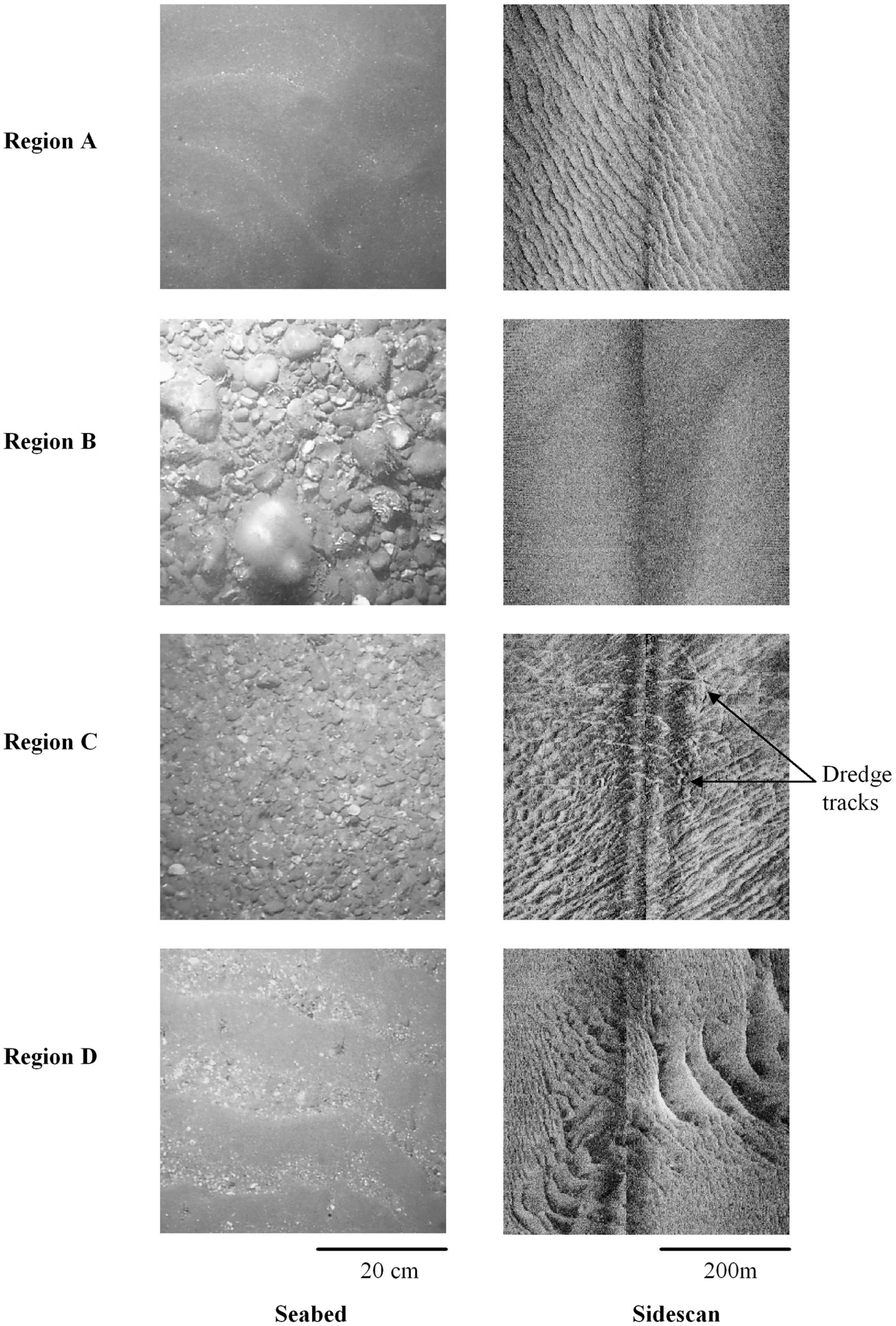

Figure 3. Examples of sidescan sonar images from the acoustically distinct regions with corresponding images of the seabed collected using the underwater video. Region A, inshore fine–medium sand <20 m; Region B, cobbles and gravel with attached epifauna—undredged Shingle Bank; Region C, disturbed gravel—dredged Shingle Bank; Region D, slightly gravelly rippled sand >20 m.

distributions of samples from within Regions A and D were also more consistent, as depicted by the tight clustering of samples in the PCA ordination (Figure 4). In contrast there was a much higher degree of particle size variability between replicate samples collected from Regions B and C, as depicted by the much wider spread of samples from these regions in the PCA ordination (Figure 4). Analysis of similarities results (Clark 1993) for

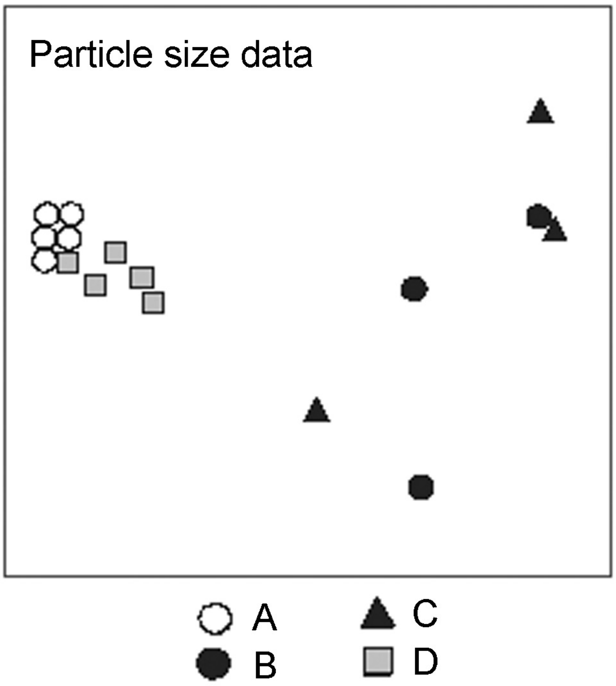

Figure 4. The PCA ordination of particle size (mean diameter in mm, sorting coefficient, % gravel, % sand and % silt/clay) distributions.

Table 1. *Summary of means and standard deviation for the numbers of species and numbers of individuals (colonial species not included in the number of individuals) from within each acoustic region from the Hamon grab and beam trawl surveys.*

	Hamon grab		2 m beam trawl	
	Mean no. taxa	Mean no. individuals	Mean no. taxa	Mean no. individuals
Region A	15(±8.2)	82(±60.1)	21(±2.3)	183(±34.0)
Region B	50(±6.4)	132(±12.9)	34(±4.0)	255(±103.6)
Region C	21(±12.7)	34(±23.1)	31(±2.3)	184(±7.6)
Region D	22(±5.4)	38(±12.5)	26(±6.1)	268(±139.0)

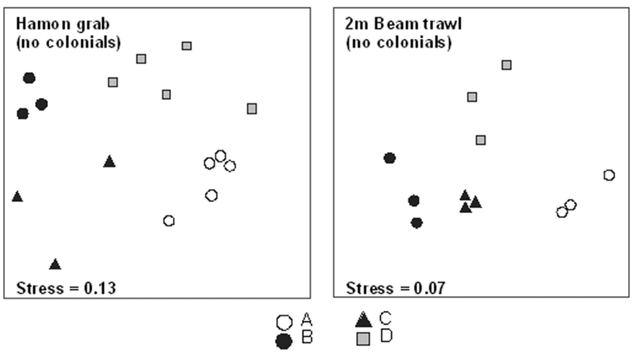

Figure 5. The MDS plots for macrofaunal assemblages from the Hamon grab and beam trawl surveys. All taxa except colonials included; data were 4th root transformed.

particle size data between samples from the four acoustic regions revealed that all regions were statistically distinct from one another, with the exception of Regions B and C. However, in terms of seabed morphology, Region C was visually and acoustically distinct from Region B, and dense dredge tracks were clearly visible on the sidescan sonar record within this region (Figure 3).

Biological data interpretation

A total of 172 taxa was identified from the 16 Hamon grab samples collected from across the survey area. There was a high degree of variability in the mean number of taxa between regions, with the undredged Shingle Bank (Region B) supporting a higher number of taxa than the dredged Bank or surrounding sandy regions (Table 1). Similarly, the undredged Shingle Bank (Region B) also supported the highest number of individuals. A total of 91 taxa was identified from the beam trawl survey. Patterns were similar to the Hamon grab data set, with the undredged Shingle Bank (Region B) supporting the highest number of individuals and taxa compared with the other regions (Table 1). Mean numbers of individuals and taxa were not markedly lower in the vicinity of the dredging (Region C) compared with the undredged Shingle Bank. However, the figures do not reflect the abundance of colonial organisms such as the soft coral *Alcyonium digitatum* and the bryozoan *Flustra foliacea*, which were notably much more abundant in Region B than Region C.

Grouping of replicate samples from each acoustic region from both the Hamon grab and beam trawl surveys is clearly visible (Figure 5). Analysis of similarities revealed that there were significant differences in macrofaunal assemblage structure between all acoustic regions, with the exception of Regions C and B from the beam trawl data.

Biotopes

The community groupings were explored further using SIMPER. Results revealed that the average similarity between replicate samples collected within an acoustic region was relatively low, particularly for the Hamon grab data, and that characterizing species from each acoustic region identified from the Hamon grab survey were unsurprisingly very different from those identified from the beam trawl survey (Tables 2 & 3).

Biotope A: *Shallow water, polychaete dominated fine sand*

The inshore area of the site (Region A), consisting of shelly sand in which polychaete tubes were visible on the underwater video footage, was identified as a discrete biotope. The species composition was characterized by polychaete worms such as *Spiophanes bombyx*, *Magelona johnstoni*, *Nephtys cirrosa* and *Aphrodita aculeata*. Burrowing amphipods of the genus *Bathyporeia* were present as was the sand goby *Pomatoschistus minutus*.

Biotope B: *Coarse gravel with attached epifauna*

Region B was the undredged region of Hastings Shingle Bank. There was an abundance of attached epifauna: in particular, the soft coral, *Alcyonium digitatum*, and the bryozoan *Flustra foliacea* distinguished this biotope from

Table 2. *Results from SIMPER analysis of Hamon grab data (all taxa excluding colonial species, 4th root transformed), listing the main characterizing species from each acoustically distinct region. Average abundance, similarity percentage, and cumulative similarity percentage for each species and the overall average similarity between replicate samples from within each region are listed.*

Acoustic region		Average abundance	%	Cumulative %	Average similarity
A	*Spiophanes bombyx*	18.2	25.0	25.0	
	Magelona johnstoni	20.8	23.6	48.6	42.3%
	Nephtys cirrosa	2.4	17.6	66.3	
	Bathyporeia gracilis	10.2	16.0	82.3	
B	*Pomatoceros triqueter*	17.7	10.6	10.6	
	Ascidiacea	11.7	8.5	19.1	
	Echinocyamus pusillus	5.0	7.7	26.8	
	Lumbrineris gracilis	5.0	7.6	34.4	43.6%
	Aonides paucibranchiata	2.7	6.5	41.0	
	Caulleriella alata	2.3	6.5	47.5	
	Scalibregma inflatum	2.0	5.8	53.4	
	Glycea lapidum	2.0	5.8	59.2	
	Poecilochaetus serpens	1.7	5.5	64.7	
	Syllis (Type B)	1.0	5.5	70.2	
C	*Caulleriella alata*	4.3	55.9	55.9	16.5%
	Scolelepis squamata	1.3	18.8	74.7	
	Ampelisca spinipes	2.7	13.7	88.4	
D	*Lumbrineris gracilis*	3.4	22.8	22.8	
	Nephtys cirrosa	2.6	13.4	36.2	27.2%
	Spisula elliptica	1.6	11.0	47.3	
	Eurydice pulchra	0.8	10.7	58.0	

Table 3. *Results from SIMPER analysis of beam trawl data (all taxa excluding colonial species, 4th root transformed), listing the main characterizing species from each acoustically distinct region. Average abundance, similarity percentage, and cumulative similarity percentage for each species and the overall average similarity between replicate samples from within each region are listed.*

Acoustic region		Average abundance	%	Cumulative %	Average similarity
A	*Pomatoschistus minutus*	54.3	13.3	13.3	
	Pagurus bernhardus	25.7	11.6	24.9	
	Aphrodita aculeata	21.0	11.2	36.1	66.4%
	Pontophilus sp.	14.0	10.8	46.9	
	Hinia sp.	15.7	9.4	56.3	
	Buglossidium luteum	11.7	8.5	64.8	
	Callionymus sp.	4.7	7.7	72.5	
	Echiichthys sp.	4.7	7.1	79.6	
B	*Psammechinus miliaris*	101.0	10.9	10.9	
	Pagurus bernhardus	27.0	10.4	21.2	
	Ophiura albida	19.3	8.8	30.0	
	Buccinum sp.	7.3	7.2	37.2	56.2%
	Macropodia sp.	5.7	6.4	43.7	
	Nudibranchia	13.3	6.2	49.8	
	Chlamys sp.	4.3	5.9	55.7	
	Pisidia sp.	6.7	5.9	61.6	
	Pomatoschistus minutus	2.7	5.7	67.3	
	Metridium senile	4.3	5.2	72.5	
C	*Pagurus bernhardus*	31.3	9.2	9.2	
	Hinia sp.	20.0	8.1	17.3	
	Pomatoschistus minutus	21.0	7.5	24.8	68.4%
	Chlamys sp.	8.7	6.9	31.7	
	Macropodia sp.	8.0	6.4	38.1	
	Galathea sp.	7.0	6.4	44.5	
	Liocarcinus sp.	11.7	6.4	50.9	
	Buccinum sp.	6.3	6.2	57.1	
D	*Pagurus bernhardus*	66.3	16.62	16.6	
	Ophiura albida	85.7	14.44	31.1	
	Liocarcinus sp.	14.0	10.61	41.7	52.5%
	Ophiura ophiura	25.3	10.27	51.9	
	Crangon allmanni	14.3	7.87	59.8	
	Pomatoschistus minutus	4.7	7.84	67.6	
	Macropodia sp.	4.7	7.61	75.2	

the others found within the study area. Other characterizing species included the sea urchin *Psammechinus miliaris*, the sea anemone *Metridium senile*, the hydroid *Sertularia* sp., the serpulid polychaete *Pomatoceros triqueter* and the encrusting bryozoan *Schizomavella* sp.

Biotope C: *Disturbed (dredged) sandy gravel*

Region C was the dredged area in the middle of the Shingle Bank, surrounded by Region B. The gravel within this region was sandier and therefore less coarse than that of Region B, and there were fewer sightings of large epifaunal species on the underwater camera footage from this area. This was confirmed by a marked absence of many of the sessile epifaunal species in the grab and trawl data that were abundant in biotope B. Whelks of the genus *Hinia* sp. were one of the characterizing species of biotope C.

Biotope D: *Deeper water, coarse sand with* Ophiura ophiura

The sediment within Region D was mainly sand with low proportions of gravel in some areas, and the particle size distribution was similar to that of Region A. However, the biotic component of this region was distinctly different, with fewer polychaete species, although the polychaete worms *Nephtys cirrosa* and *Spiophanes bombyx* were present as they were in Region A. The brittle stars *Ophiura albida* and *Ophiura ophiura* were identified as characterizing species from this habitat.

DISCUSSION

The Hastings Shingle Bank and surrounding seabed have been well studied for a number of years due to the interest in the site for aggregate extraction (Shelton & Rolfe, 1972; Kenny, 1998). The location has also been sampled as part of broader-scale benthic surveys in the English Channel (Holme & Wilson 1985; Sanvicente-Anorve et al., 1996). In the current study, four biotopes were identified from an area 12×4 km which encompassed the Shingle Bank and parallels can be drawn between these and assemblage types described in the past.

The undredged region of the Shingle Bank was dominated by the soft coral, *Alcyonium digitatum* and the bryozoan *Flustra foliacea* attached to coarse deposits of cobbles, pebbles and gravel. These two characterizing species, amongst others, have been reported in this vicinity in previous surveys (Shelton & Rolfe, 1972). Holme & Wilson (1985) describe several epifaunal assemblages from the central region of the English Channel which show a degree of similarity to those found at the Hastings study site. They document three sub-types of an assemblage (Type B) associated with hard surfaces of rock, cobbles and pebbles which are subjected to varying degrees of tidal scour by sand and periodic smothering, namely:

- Subtype B-1 'Well developed faunal assemblage with *Polycarpa violacea* assemblage' (Holme & Wilson 1985). This is described as a relatively stable, rich and varied fauna associated with pebbles, cobbles and rock outcrops, affected periodically by sand scour.
- Subtype B-2 'Impoverished *Polycarpa violacea–Flustra foliacea* assemblage' (Holme & Wilson 1985). This assemblage is found on similar hard substrates as subtype B-1, but is subjected to considerable sand scour and periodic submergence by thin layers of sand.
- Subtype B-3 'Impoverished *Balanus–Pomatoceros* assemblage' (Holme & Wilson 1985). This assemblage is characteristic of hard substrates subjected to severe scour and deep submergence by sand or gravel. The fauna is therefore restricted to fast-growing colonizers which can rapidly settle and establish themselves in the short periods when conditions are favourable.

The undredged region appears similar in terms of fauna and physical habitat to subtype B-1 and B-2. The dredged Shingle Bank (Biotope C), which consists of a sandier substrate, and dredge tracks in-filled with sand, shows similarity to the subtype B-3. Kenny (1998) draws similar comparisons between the assemblages reported by Holme & Wilson (1985) and those he identified within the region of Hastings Shingle Bank during an environmental survey of the areas licensed for aggregate extraction. Shelton & Rolfe (1972) describe a rich fauna on the Shingle Bank but did not identify any impoverished regions. This can be explained by the fact that trailer dredging for aggregates did not begin 'in earnest' at the licensed sites until 1988, and it is likely that the rich fauna found on the undredged Shingle Bank (Biotope B) originally extended over the entire area of the Bank.

Studies by Shelton & Rolfe (1972) and Kenny (1998) also report the presence of sandier deposits to the north and south of the Shingle Bank, in agreement with the current study. However, these previous studies focused their survey effort within the immediate vicinity of the Shingle Bank, or extended surveys in a south-west–north-east direction parallel with the prevailing tidal currents. There is limited previous data regarding the benthic fauna to the north and south of the bank.

Habitat boundaries between acoustically distinct regions within the study site were relatively clear. Moreover, the acoustic regions themselves appeared to coincide with discrete assemblages identified by ground-truthing. However, similar studies attempting to map the spatial distribution of habitats and assemblages elsewhere have indicated that a close association between the two does not always exist (Basford et al., 1989; Dewarumez et al., 1992; Brown et al., 2002). This concept of discrete communities vs continua is discussed by Brown et al. (2002), and the study site described in the current study appears to fall into the latter category, displaying very distinct faunal differences which appear to coincide with clearly discernible habitat boundaries.

Other factors, such as sediment characteristics, are thought to have a greater influence on assemblage structure at more localized scales, such as those encountered in the current study (Eleftheriou & Basford, 1989; Seiderer & Newell, 1999). Substratum types can often show discontinuities across a region which may give rise to distinct boundaries between neighbouring assemblages. The use of sidescan sonar in the current study enabled such boundaries to be identified and mapped. Designing subsequent biological surveys around the acoustically distinct regions determined from the sidescan sonar data made it possible to test whether discrete assemblages existed within these boundaries. However, the lack of clearly definable boundaries between adjacent habitats can cause major problems when attempting to produce high-resolution seabed maps due to difficulties in determining where demarcation lines should be drawn.

The mapping approach also proved very successful in identifying anthropogenic disturbance at the seabed from aggregate dredging. Dredging tracks were clearly identifiable from the sidescan data, and an impoverished fauna was recorded from within the disturbed area compared with that from the surrounding undisturbed gravel bank.

The survey approach adopted in the current study, using a combination of sidescan sonar, video, grab and trawl, has led to a detailed understanding of the spatial distribution of habitats and assemblages within the region. The use of a swath acoustic system such as sidescan sonar allows 100% coverage of the survey area to be achieved. This in turn increases the accuracy at which habitat boundaries can be drawn across the area, which ultimately increases the confidence of the final biotope map when compared with other mapping approaches.

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Mapping seabed biotopes using sidescan sonar in regions of heterogeneous substrata: Case study east of the Isle of Wight, English Channel

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Abstract

As part of a wider research programme to investigate the utility of acoustic remote sensing techniques for mapping seabed biotopes, an area of seabed to the east of the Isle of Wight, some 12x4 km in size, was surveyed using sidescan sonar. The study site was selected due to the high degree of sediment heterogeneity within the region. A mosaic of the sidescan sonar data was produced to provide a 100% spatial coverage map of the study site, which was subsequently divided into acoustically distinct regions, and each region ground-truthed using a suite of physical sampling and optical techniques. The main sampling tools, a $0.1m^2$ Hamon grab and a drop-camera system, were used to characterise the benthic communities and sediment characteristics within each acoustically distinct region. Relationships between acoustic regions, physical habitat characteristics and assemblages were investigated using a range of statistical techniques. Results from these analyses were used to identify discrete biotopes. Statistically distinct assemblages were identified within a number of the acoustic regions, although the high degree of sediment heterogeneity proved problematic in identifying discrete boundaries to physical habitats. Problems associated with delineating the spatial distribution of biotopes over regions of seabed comprising differing degrees of habitat/sediment complexity are discussed.

1. Introduction

The use of remote sensing technologies in marine environmental monitoring and research has increased substantially in recent times. [1-15] This increase can be mainly attributed to improvements in acoustic systems in the 1990s, in particular with swathe and multibeam systems as a result of increased digital processing power offered by modern computers. This has led to high-resolution and affordable systems entering the market place. Although the outcomes of many of the above referenced studies are, in general, encouraging, the approaches have not yet reached the stage of uncritical, routine application.

Whilst it has been demonstrated that a combination of acoustic and remote sampling techniques can be used to map the spatial distribution of seabed habitats in regions of homogeneous substrata within clearly definable boundaries, [1, 11, 15] mapping regions of heterogeneous substrata is far more difficult. Much of the seabed surface around the England and Wales coastline is comprised of coarse material. Such areas often consist of a mixture of substratum types and habitats displaying a high degree of spatial complexity which makes them difficult to map.

Where these coarse substrate deposits are present in sufficient quantity, are of the right consistency, and are accessible to commercial dredgers, they may be exploited as a source of aggregate for the construction industry (to supplement land-based sources) or as a source of material for beach nourishment. It is therefore often desirable to map such regions for the purpose of environmental monitoring and management. The study presented in this paper was conducted as part of a wider study, funded by the UK Department for Environment, Food and Rural Affairs (DEFRA), which aimed to evaluate the utility of a number of acoustic systems for mapping seabed biotopes in areas of coarse substrata.

This paper addresses the issue of producing high-resolution habitat/biotope maps in regions of complex, heterogeneous seabed and a case study is presented from an area of seabed in the English Channel to the east of the Isle of Wight.

2. Methods – Acoustic surveys

Sidescan sonar survey

An intensive survey of the study site to the east of the Isle of Wight (Fig 1) was carried out in 1998 using an EG&G DF1000 analogue sidescan sonar system with a Triton Isis logging system. Delphmap post-processing software was used to mosaic the imagery and classify texturally different regions. The system was operated on a 300m swathe range, and survey lines were spaced at 300m intervals in a north-south orientation in order to image 100% of the survey area. A drop-camera frame fitted with an underwater video camera and light was

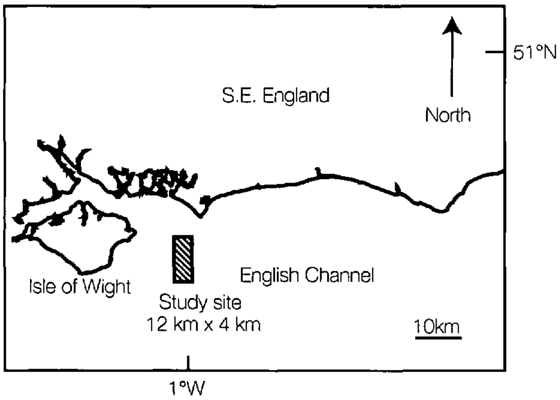


Fig 1: Location of the survey sites, in the English Channel east of the Isle of Wight

deployed at a number of stations across the survey area in order to provide visual ground-truth data to aid interpretation of the sidescan sonar data set. Echosounder data was also recorded at the same time as the sidescan sonar data to provide crude bathymetric data from across the study site.

Data interpretation

Seabed features (sand ripples, bedrock outcrops etc) and an indication of the sediment characteristics ('soft' or 'hard' sediments) could be identified from the sidescan sonar mosaic and the presence of these features/characteristics was confirmed through the underwater video data collected at the ground-truth stations. Using this approach, the study site was broadly divided into 'acoustically distinct' regions based on the textural backscatter information derived from the output of the sidescan sonar. These regions formed the basis for the design of subsequent biological and sedimentological surveys.

3. Methods – Biological surveys and ground-truthing

Survey design and sampling

The main sampling tool was a 0.1m^2 Hamon grab[16] fitted with a video camera and light. This was the preferred type of sampling gear due to its ability to collect samples on coarse, unconsolidated sediments. The grab was fitted with a video camera in order to record an image of the seabed adjacent to the collection bucket of the grab, thus providing information about the undisturbed surface of the substratum at each sampling station. Sampling stations were randomly positioned within each acoustic region and the number of stations within each region was linked to the size of the area.

A total of 25 Hamon grab samples were collected across the study site (July/August 2000). Following estimation of the total volume of each grab sample, a 500ml sub-sample was removed for laboratory particle size analysis. The remaining sample was washed over 5mm and 1mm square mesh sieves to remove finer sediments. The retained macrofauna were fixed in 4-6% buffered formaldehyde solution (diluted with seawater) for laboratory identification and enumeration.

A drop-camera frame fitted with a video camera and lights was also deployed at a number of stations to obtain additional visual ground-truth data from each of the acoustic regions. The camera system was suspended above the surface of the seabed as the vessel was allowed to drift. Deployments were made during tidal velocities of less than 1 knot in order to achieve good quality video footage.

Sample processing

In the laboratory, macrofauna samples were washed with freshwater over a 1mm square mesh sieve to remove excess formaldehyde solution. Samples were then sorted and the specimens placed in jars or petri-dishes containing a preservative mixture of 70% ethanol, 10% glycerol and 20% tap water. Specimens were identified to species level, as far as possible, using standard taxonomic keys. The number of individuals of each species was recorded with colonial species recorded as present or absent. For each positive identification a representative specimen was retained in order to establish a reference collection.

The sediment sub-samples from each grab station were analysed for their particle size distributions. Samples were first wet sieved on a 500 micron stainless steel test sieve, using a sieve shaker. The <500 micron sediment fraction, along with water from the wet sieving, was allowed to settle in a bucket for 48 hours. Excess water was then removed using a vacuum pump and the fraction was washed into a sterile petri-dish, frozen for 12 hours and freeze dried. The weight of the sediment was also recorded. A sub-sample of the <500 micron freeze dried fraction was then analysed using a laser sizer.

The >500 micron fraction was washed from the test sieve into a foil tray and oven dried at ~90°C for 24 hours. It was then dry sieved for 10min on a range of stainless steel test sieves at half phi intervals, down to 1 phi. The sediment on each sieve was weighed to 0.01g and the results recorded. The results from these analyses were combined to give the full particle size distribution. The mean and sorting values were then calculated.

Data analysis

Univariate analysis

Total number of individuals (excluding colonial species) and total number of species within each acoustic region were calculated from the Hamon grab survey as summary measures of benthic assemblages. Bartlett's test was used to test for homogeneity of variance. Where the variance was not homogeneous a log transformation of the data was carried out. The significance of differences between acoustic regions was tested using one-way ANOVA. Kruskal-Wallis test was applied to test for a difference in the number of species between the regions (variance was not homogeneous for the mean number of species), and Fisher's least significant difference (LSD) multiple comparisons procedure was used to determine significant differences in the number of individuals between regions. Univariate analyses were performed using the software package STATGRAPHICS Plus (version 4).

Multivariate analysis

Associations between benthic assemblages and acoustic regions were examined using multivariate statistical methods. Sample and species associations across the survey area were assessed by non-metric multi-dimensional scaling (MDS) ordination using the Bray-Curtis similarity measure on 4th root transformed abundance data (excluding colonial taxa) using the software package PRIMER (version 4)[17]. Rare species (ie, with fewer than three individuals recorded throughout the survey area) were removed from the data set in order to reduce the variability caused by these infrequently occurring species. Removing these species was also necessary to conform to certain limitations in the total number of species which can be used during certain tests within the PRIMER (version 4) software (eg, SIMPER – see below).

Analysis of similarities (ANOSIM,[18]) was performed to test the significance of differences in macrofauna assemblage composition between samples. The nature of the groupings identified in the MDS ordinations were explored further by applying the similarity percentages program (SIMPER) to determine the contribution of individual species to the average dissimilarity between samples.

A correlation-based principal components analysis (PCA) was applied to ordinate results from the sediment particle size analysis. Prior to analysis, environmental variables were converted to approximate normality using a log $(1+N)$ transformation. Analysis of similarities (ANOSIM,[18]) was performed on particle size data to test the significance of differences in particle size composition between acoustic regions.

4. Results

Acoustic data interpretation

Examination of the sidescan sonar data revealed a complex and heterogeneous seabed. Underwater video footage revealed a high level of small-scale sediment variability, which made it difficult to establish at what scale distinct habitats should be defined. A pragmatic approach was therefore adopted; gross habitat differences were interpreted from the sidescan sonar mosaic and boundaries were drawn at a scale with which ground-truthing equipment (eg, grab, drop camera) could be accurately placed on the seabed within the boundaries of each discrete region. Ultimately, the site was divided into five acoustic regions (labelled A, B, C, D and E), whilst realising that there was a high level of substratum variability and patchiness within each acoustic region.

There were some similarities between regions A, B and C in the north of the survey area: Region A was interpreted as an area of hard substrata (rocks and cobbles) covered with a thin veneer of softer substrata (sand/silt); Region B was similar, but had a much higher proportion of soft substrata showing signs of sediment ripples, and appeared to be associated with a topographical depression; Region C had a higher proportion of hard, reflective substrata (out-cropping rock/boulders), with fewer patches of soft substrata. In contrast, the two regions in the south of the survey area consisted predominantly of soft substrata and showed a high degree of small-scale sediment heterogeneity. Region D appeared to be a patchwork of substrata (sand, gravel, cobbles and outcropping bedrock). Region E appeared similar to region D but was covered by large areas of soft sediment veneers.

These acoustic interpretations were confirmed from the drop-camera video ground-truthing. The acoustic regions and the position of Hamon grab samples are

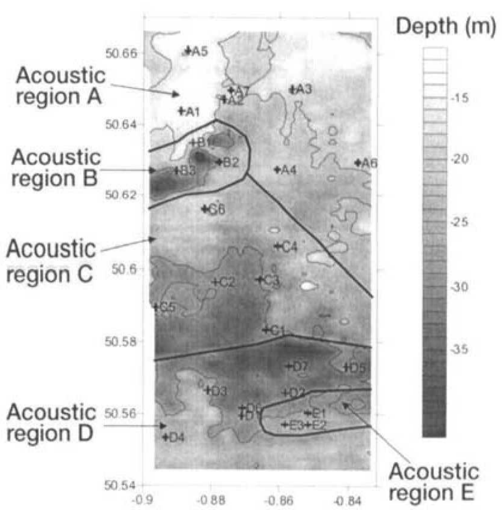


Fig 2: Bathymetric plot of the Isle of Wight survey area showing the five acoustically distinct regions (A, B, C, D, E) determined from the sidescan sonar data, and locations of the Hamon grab sampling stations. Depth contours have been plotted from echosounder data collected at the site

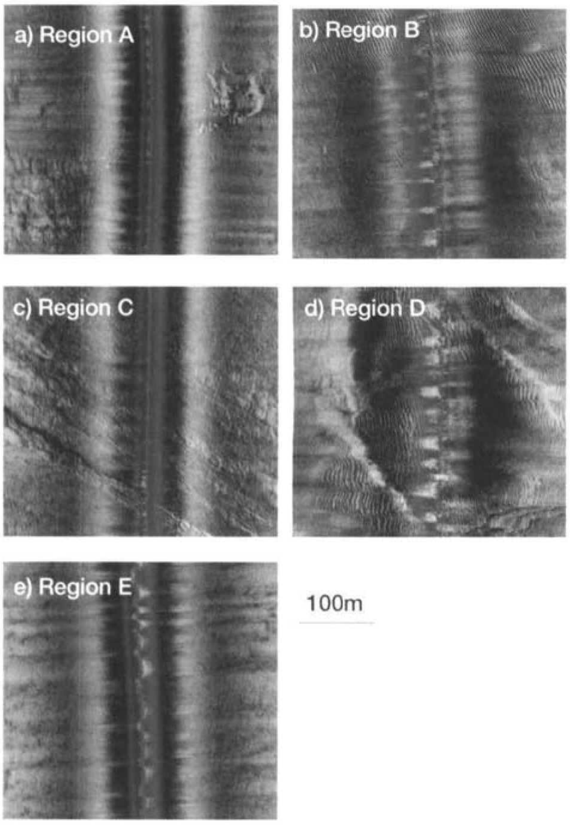


Fig 3: Examples of the sidescan sonar data from: a) Region A showing hard substrata (rocks and cobbles) covered with a thin veneer of softer sediment [sand/silt]; b) Region B showing a rippled soft sediment veneer (sand/silt) overlying hard substrata [rocks and cobbles]; c) Region C showing a silt/sandy veneer over a rock/cobble pavement with patches of surface cobbles and boulders; d) Region D showing small-scale substratum patchiness (sand, gravel, cobbles and outcrops of bedrock); e) Region E showing relatively thick coarse sand veneers

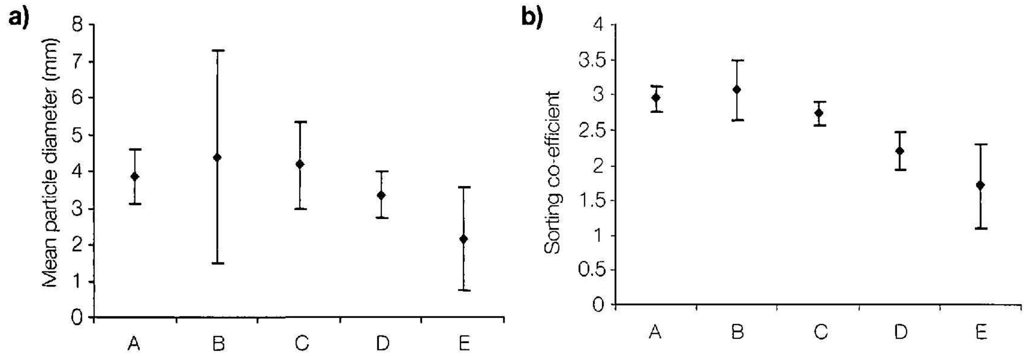
shown in Fig 2. Examples of the sidescan sonar record from each of the acoustic regions are shown in Fig 3.

Sediment characteristics and environmental variables

Particle size data (Fig 4) revealed that there was a high degree of variability in the mean particle diameter between replicate samples. This reflects the heterogeneous nature of the substrata within each acoustic region and makes it very difficult to detect discrete habitats on the basis of these data. Results from analysis of similarity tests (ANOSIM[18]) confirmed that in most cases there was no significant difference in particle size distribution between samples from each acoustic region, with the exception of regions A and B, A and E ($p<0.1$) and A and D ($p<0.05$) (Table 1).

An ordination by PCA of the particle size distributions from the grab samples is illustrated in Fig 5. There was a large degree of overlap between samples from different acoustic regions, and there was no obvious grouping of samples from within each region (with the possible exception of a number of samples from region D).

Despite the difficulty in identifying discrete habitats on the basis of the particle size data, there did appear to be acoustic and visual differences in terms of the physical habitat characteristics between the five regions, although the variations tended to be relatively subtle. Examination of the underwater video footage revealed



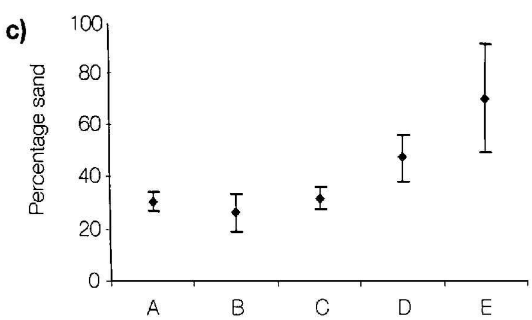


Fig 4: Sediment particle size characteristics from Hamon grab samples collected from acoustic regions A-E: a) Mean particle size in mm; b) Sediment sorting coefficient; c) Percentage sand. Individual variance shown as error bars

	A	B	C	D
B	*			
C	n.s.	n.s.		
D	**	n.s.	n.s.	
E	*	n.s.	n.s.	n.s.

Table 1: Analysis of similarity for particle size characteristics (mean diameter in mm, sorting coefficient, % gravel, % sand and % silt/clay) between acoustically distinct regions (n.s. = not significant; * Significant at p < 0.1; ** Significant at p < 0.05)

that regions A, B and C tended to be slightly muddy in nature and this elevated percentage of fine material was evident in some of the particle size analyses, but was not found to be statistically significant.

The substratum in region B also consisted of high numbers of *Crepidula* shells. In contrast, the two regions in the south of the study site, regions D and E, comprised coarse material and out-cropping bedrock overlain with areas of sand veneers (region D), some of which were extensive in their coverage (region E). Substrata in many of the regions were also consolidated, and there appeared from the video footage to be a well developed epifaunal community.

Biological data interpretation

A total of 338 taxa were identified from the 25 Hamon grab samples collected from across the survey area. Mean numbers of species and individuals per grab sample from each acoustic region are shown in Fig 6. The mean number of species and individuals across the sites ranged from a mean of 9 species and 12 individuals per grab sample in region E, to 58 species and

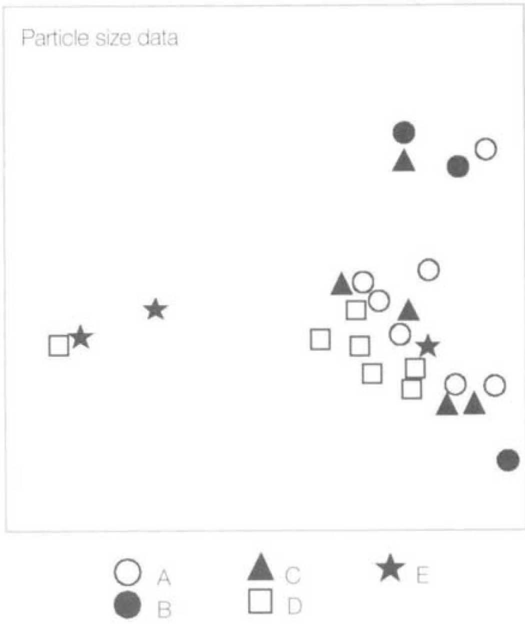


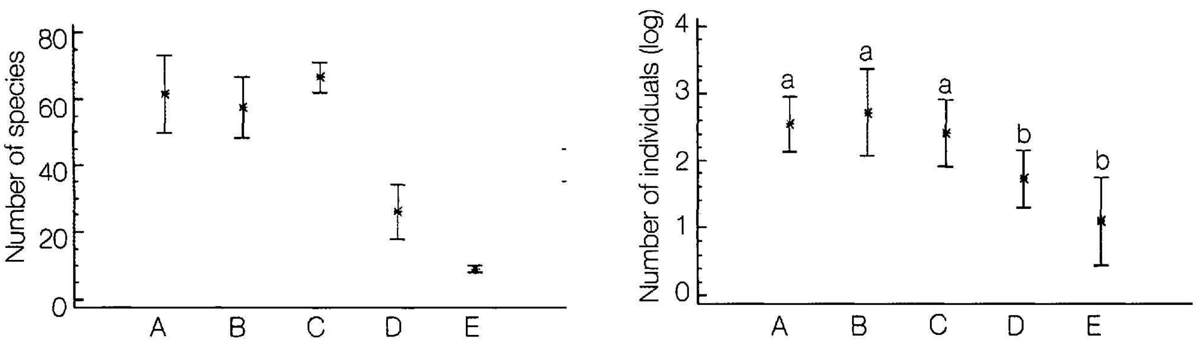
Fig 5: PCA ordination of particle size characteristics (mean diameter in mm, sorting coefficient, % gravel, % sand and % silt/clay)

759 individuals per grab sample in region B, although the high numbers of individuals can be attributed mainly to a few taxa which were highly abundant at several of the sampling stations (eg, *Balanus crenatus*, *Sabellaria spinulosa* and *Crepidula fornicata*).

Univariate tests on the Hamon grab data showed a statistically significant difference between the median number of species from certain acoustic regions, with regions A, B and C having higher numbers of species than regions D and E (variance was not homogeneous for the mean number of species, therefore the Kruskal-

Fig 6: Summary of means with standard deviations (number of species) or 95% pooled confidence intervals (number of individuals) from within each acoustic region (Hamon grab survey). Values for the number of individuals labelled with the same letter are not significantly different from one another at p<0.05, following application of Fisher's LSD multiple comparison procedure

Acoustic Region	A	B	C	D	E
Mean No. Species	61	58	67	26	9
Mean No. Individuals	573	759	287	105	12



	A	B	C	D
B	63			
C	58	59**		
D	77**	80**	76**	
E	88**	88*	90**	77*

Table 2: Dissimilarities (%) between assemblages from within acoustically distinct regions based on 4th root transformed data. * denotes significant difference at $p < 0.1$. ** denotes significant difference at $p < 0.05$

Table 3: Results from SIMPER analysis of Hamon grab data (all taxa excluding colonial and low abundance species, 4th root transformed), listing the main characterising species from each acoustically distinct region. Average abundance, percentage contribution to similarity, and cumulative percentage contribution to similarity for each species and the overall average similarity between replicate samples from within each region are listed

Acoustic Region	Taxon	Average Abundance	%	Cumulative %	Average Similarity
A	*Ampelisca spinipes*	36.57	5.42	5.42	
	Clymenura sp.	8.71	5.35	10.77	
	Nemertea	5.29	5.26	16.03	
	Lumbrineris gracilis	5.14	4.79	20.82	
	Pomatoceros lamarcki	20.86	3.85	24.66	
	Polycirrus sp.	7.57	3.61	28.27	
	Typosyllis variegata	14.00	3.26	31.53	38.76%
	Balanus crenatus	181.29	3.13	34.66	
	Elminius modestus	13.14	3.09	37.75	
	Nematoda	7.57	3.04	40.78	
	Notomastus sp.	4.14	3.04	43.82	
	Sabellaria spinulosa	46.14	3.00	46.82	
	Harmothoe impar	4.14	2.73	49.56	
B	*Ampelisca spinipes*	19.33	7.73	7.73	
	Crepidula fornicata	21.33	5.63	13.35	
	Amphipholis squamata	10.67	5.28	18.64	
	Nucula nucleus	11.00	4.91	23.55	
	Mediomastus fragilis	5.00	4.70	28.25	41.83%
	Lumbrineris gracilis	4.00	4.66	32.91	
	Caulleriella alata	3.00	4.66	37.57	
	Scalibregma celticum	4.33	4.52	42.09	
	Polycirrus sp.	2.67	4.03	46.12	
	Pisidia longicornis	6.00	4.03	50.14	
C	*Sabellaria spinulosa*	21.00	4.08	4.08	
	Pisidia longicornis	7.40	3.79	7.86	
	Sphenia binghami	12.20	3.67	11.53	
	Typosyllis variegata	6.60	3.61	15.14	
	Nemertea	4.00	3.53	18.67	
	Verruca stroemia	10.60	3.48	22.15	
	Amphipholis squamata	4.80	3.46	25.61	52.80%
	Lumbrineris gracilis	3.00	3.25	28.86	
	Clymenura sp.	4.60	3.24	32.09	
	Ampelisca spinipes	3.60	3.06	35.15	
	Anomia (juv)	1.00	2.62	37.78	
	Tricolia pullus	1.00	2.62	40.40	
	Molgula manhattensis	13.60	2.49	42.89	
	Modiolus tumida	5.00	2.38	45.27	
D	*Notomastus sp.*	2.43	18.33	18.33	
	Eulalia mustela	1.57	18.12	36.46	
	Balanus crenatus	30.57	10.50	46.96	
	Ophelia borealis	3.57	9.78	56.74	
	Nematoda	2.43	3.57	60.31	29.45%
	Polycirrus sp.	0.71	3.16	63.48	
	Travisia forbesii	0.43	3.06	66.54	
	Pseudoprotella phasma	2.14	3.03	69.57	
	Crepidula fornicata	1.14	2.64	72.21	
	Nemertea	0.71	2.57	74.78	
E	*Nemertea*	1.00	59.97	59.97	
	Ophelia borealis	1.33	21.18	81.15	22.27%
	Balanus crenatus	1.33	18.85	100.00	

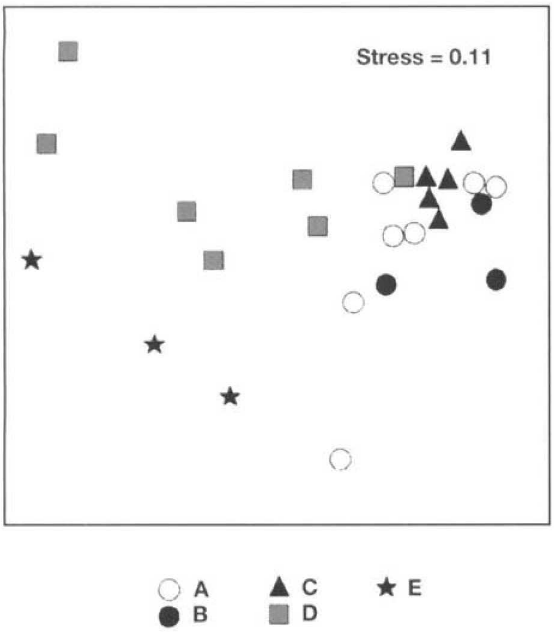


Fig 7: MDS plots for macrofaunal assemblages from the Hamon grab surveys. All taxa except colonial species included; data were 4th root transformed

Wallis test was applied to test for a significant difference between regions). Similarly, regions A, B and C had significantly higher numbers of individuals than regions D and E (Fig 6).

The output from a non-metric multidimensional scaling (MDS) ordination of data from the Hamon grab survey is shown in Fig 7. There were no clear patterns in community structure. Replicate samples from most acoustic regions were not tightly clustered in the ordination, and there was a high degree of overlap between regions. However analysis of similarities (ANOSIM) did reveal the presence of statistically distinct assemblages in a number of acoustic regions (Table 2). Regions A, B and C had similar assemblage structures, which were statistically distinct from regions D and E which were also similar in terms of assemblage structure. This is reflected in the ordinations; there is a degree of separation between regions A, B and C and regions D and E.

Biotopes

The community groupings were explored further using the similarity percentages routine SIMPER, and results revealed characterising species from each of the acoustic regions. Average similarity between replicate samples collected within an acoustic region was relatively low, particularly where the substratum was very patchy (eg, regions D and E). Similar characterising species were identified from regions A, B and C, and from regions D and E (Table 3). These results, along with information derived from the sidescan sonar

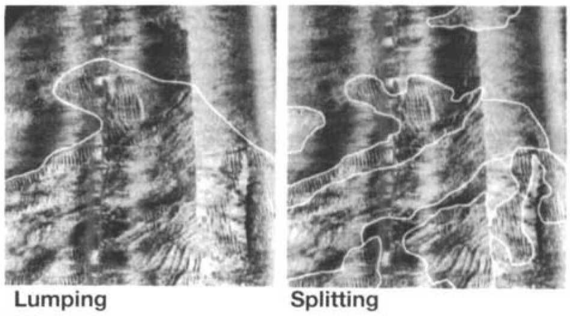


Fig 8: Alternative demarcation of boundary lines between acoustic regions: 'lumping' vs 'splitting'

mosaic and underwater video and photographic material, were used to derive biotopes, and are listed below.

Biotope A/B/C: Slightly muddy, sandy gravel with epifauna encrusted cobbles

A common feature of the regions within this biotope was the high content of fine sediments and a variable proportion of cobbles with abundant epifaunal growth, as revealed by the underwater camera. The species composition of region A showed strong similarities with both regions B and C; however the species compositions of regions B and C were less similar to each other. This suggests that region A may represent an intermediate habitat incorporating elements of both B and C. The amphipod *Ampelisca spinipes* and the polychaete worm *Lumbrineris gracilis* were important characterising species of the benthic assemblages for all three regions.

Biotope D/E: Consolidated gravel/rock covered by sand veneers with Ophelia borealis

Regions D and E were highly variable in terms of their physical habitat. The underwater camera revealed that region D consisted of large areas of sandy gravel. Within region D there were also areas with coarse sand intersected by slight depressions containing gravel and cobbles. Region E was similar to the latter, being mostly comprised of coarse sand with a few cobbles. The variability in species composition between replicates within these two regions reflects the physical heterogeneity of the two acoustic regions. The polychaete worm *Ophelia borealis* was, however, an important contributor to similarity in both regions D and E.

5. Discussion

A number of previous studies have examined the spatial distribution of benthic species, communities and habitats within the English Channel,[19, 20, 21, 22, 23] and several of these investigations have noted the rough topography and complex nature of the seabed in the Isle of Wight area. It was clear from the sidescan sonar mosaic produced in the current study that the survey area consisted of a range of sediment types with a high degree of spatial heterogeneity. This proved problematic when attempting to define habitat boundaries around regions of acoustically complex substrata.

A number of habitat mapping studies have illustrated that sidescan sonar can be used to identify and delineate discrete physical seabed habitats within which distinct benthic assemblages occur. [1, 11, 15] These studies were carried out in areas where there tended to be definite boundaries between neighbouring homogeneous regions of dissimilar sediments. This obviously facilitates the demarcation of boundaries lines, and increases the likelihood of discrete assemblages occurring within each acoustic region. However, the small-scale patchiness of sediments within the current study complicated the placement of boundary lines and made division of the area into discrete regions more difficult.

The process of selecting at what scale habitat boundaries should be delineated, as illustrated in Fig 8, has important implications for the utility of the outcome in an environmental management context. Two difficult choices present themselves: the first requires generalisation ('lumping') in order to define regions that will, in all probability, contain variable substrata within some broader basis for division (eg, depth); the second, which may be viewed as more scientifically credible, requires division ('splitting') into as many regions as are formally necessary or possible in accordance with observed substratum variability. Both of these approaches have drawbacks. A possible effect of 'lumping' is that boundaries may be drawn which are more of a human artefact than a true ecological separation. However, 'splitting' may result in regions that cannot be accurately ground-truthed.

The scale at which boundaries were defined in the current study was ultimately determined by the limits of the ground-truthing accuracy. This meant that regions were defined that were inherently patchy encompassing a range of sediment types (ie, 'lumping'). Whilst it would have been possible to map the small-scale changes in sediments at a higher resolution based on the sidescan sonar data, it would have proven more difficult to ground-truth them accurately with the techniques available (grabs and camera). Fig 8 illustrates this point, showing the highest resolution at which the surficial sediments could have been delineated from a section of the sidescan sonar mosaic of the study site ('splitting'), and the actual boundaries that were drawn over the same area ('lumping') as limited by the positional accuracy to which the ground-truthing equipment could be deployed.

The high level of sediment heterogeneity within the survey area was reflected in the infaunal data. The average similarity of infaunal communities from grab samples collected from within each region was low (22-52%, Table 3), suggesting a high level of spatial heterogeneity in species distributions over the survey site. It is likely that the high spatial heterogeneity of the biota is linked to the high degree of sediment spatial heterogeneity.

It is possible that discrete communities do exist at a higher resolution within discrete patches of each type of substrata identified from the sidescan sonar data (Fig 8 – 'Splitting'). However, limitations in terms of the positional accuracy of the ground-truthing methodology prevented the identification and mapping of biotopes at this resolution. Despite this limitation, discrete biological assemblages were still identified across the study site at the broad scale at which the acoustic regions were defined (Fig 8 – 'Lumping').

Two biotope classes were recognised following statistical analysis of the macrofaunal data: one in the north of the study site encompassing three of the acoustic regions (A, B and C) consisting of a slightly muddy, sandy gravel with epifauna encrusted cobbles; and one in the south of the study area encompassing two of the acoustic regions (D and E) consisting of consolidated rock and gravel covered by sand veneers inhabitated by the polycheate worm *Ophelia borealis*. It is likely that these broad biotopes represent a cross section of the biological component from a number of small-scale sediment patches sampled by the Hamon grab from within each broad acoustic division. The two discrete biotopes may ultimately have been recognised as a result of gross habitat differences between the north and south of the study site.

Whether or not discrete assemblages exist within the small-scale sediment patches (highlighted in Fig 8) is a matter for debate. Previous habitat mapping studies attempting to map the spatial distribution of habitats and assemblages elsewhere have suggested that it may not always be possible to identify statistically discrete benthic assemblages associated with acoustically distinct regions. [13, 24, 25, 26] In these studies a close association between acoustically distinct substrata and discrete benthic assemblages did not always exist, and examples of gradations of change rather than distinct boundaries are given.

The concept of discrete communities versus continua has long been debated. Glémarec[27], after reviewing the evidence of a number of earlier studies, concluded that for level-bottom communities gradual changes in the composition of the fauna, rather than sharp discontinuities, are the norm. However, he recognised that in order to produce maps it is necessary to draw demarcation lines, and as a result communities are defined which relate to the peaks of frequency (or noda) within the continuous gradient of faunal composition. Positioning of boundaries within the current study are based on a similar principle, encompassing regions of subtly different assemblages caused by gradations in species distributions as a result of the heterogeneous nature of the substrata.

It is important for environmental managers, especially those concerned with biotope definition for conservation or environmental management decisions, to recognise that even using state-of-the-art mapping approaches, the inherent complexity of some areas defies neat classification other than into general broad categories. While this may be viewed by some as a scientifically unsatisfactory outcome, it is precisely this complexity (in the Isle of Wight case, reflected in substrate heterogeneity) which characterises the region and explains the high diversity of the indigenous biota.

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Paper #15

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ESTUARINE COASTAL AND SHELF SCIENCE

Preliminary observations of the effects of dredging intensity on the re-colonisation of dredged sediments off the southeast coast of England (Area 222)

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Abstract

The re-colonisation of a site used for the extraction of sand and gravel for ca. 25 years off the southeast coast of the UK was examined 4 years after the cessation of dredging. Effects of different levels of dredging intensity on the rate of macrofaunal re-colonisation were investigated. Values of abundance and total numbers of species were significantly lower ($p < 0.05$) in an area most recently exposed to the highest level of dredging intensity compared with samples taken from an area of low intensity, and those from a reference site. Differences between previously dredged sediments and the reference location were due to the reduced abundance of a range of macrofaunal species characterising nearby sediments. Multivariate measures of community structure also indicated that there were significant differences ($p < 0.01$) between the macrofaunal assemblages in the areas exposed to different dredging intensities. Sediment from the area exposed to the highest dredging intensity contained proportionally more sand than other sampled sediments. The extent to which dredging intensity contributed to these differences was difficult to determine owing to the absence of any baseline data. Despite this, univariate and multivariate analyses indicated a strong relationship between macrofaunal community structure and dredging intensity at this site. Correlation analyses also demonstrated that the predominant influence on the macrofaunal community was that of the level of dredging that took place in 1995, the last year that the licensed site was dredged heavily.

Preliminary observations indicated that the fauna remained in a perturbed state some 4 years after cessation of dredging. Therefore, relatively rapid 'recovery' rates, commonly cited as 2–3 years for European coastal gravelly areas, should not be assumed to be universally applicable. Implications for the future management and scientific study of marine aggregate extraction arising from preliminary observations on the physical and biological status of this site are discussed together with the options for selecting a reference site in the absence of baseline information.

Keywords: aggregate extraction; dredging; recovery; macrofauna; re-colonisation; intensity; North Sea

1. Introduction

Industrial exploitation of the UK marine aggregate resource peaked in 1989, and has remained relatively steady in recent years at around 23 Mt per annum from around the England and Wales coastline (Crown Estate records, unpublished). Concerns over the effects of marine sand and gravel extraction on the environment and fisheries have grown over time, and this is particularly the case at localities off the eastern and southern English coastlines, which are characterised by the occurrence of a number of dredging licences in close proximity.

Studies of benthic re-colonisation in the aftermath of dredging in UK waters and elsewhere are limited, and are largely confined to experimental circumstances (e.g. Cressard, 1975; van Dalfsen et al., 2000; Desprez, 2000; Desprez & Duhamel, 1993; Jewett, Feder, & Blanchard, 1999; Kenny & Rees, 1994, 1996; Kenny, Rees,

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Greening, & Campbell, 1998; Millner, Dickson, & Rolfe, 1977; Shelton & Rolfe, 1972; Van Moorsel, 1993, 1994; Van Moorsel & Waardenburg, 1990, 1991). Investigations of the physical and biological status of the licensed areas in UK at various times following cessation of commercial dredging are very limited (e.g. Millner et al., 1977), and so the judgements with regard to the likely progress towards environmental restoration and the time-scales involved continue to be based on predictions rather than real data.

Most dredged sediments will initially be rapidly re-colonised by benthic species following the cessation of dredging (Cressard, 1996; Hall, 1994; López-Jamar & Mejuto, 1988; McCauley, Parr, & Hancock, 1977; Swartz, DeBen, Cole, & Bentsen, 1980). While assessing 'recovery' rate, it is important to draw a distinction between re-colonisation, which is the settlement of new recruits from the plankton or immigration of adults from outside the area (i.e. the start of the process of restoration), and restoration, which can be considered as the return of community structure. An indication of the level of restoration can be gained by comparing the attributes of a community from the disturbed area with those from reference sites or with 'baseline' (i.e. pre-dredging) conditions. Community attributes that are most commonly investigated in this respect include numbers of species, abundance, biomass and the age structure of populations (Bonsdorff, 1983; van Dalfsen et al., 2000; Desprez & Duhamel, 1993; Essink, 1997, Jewett et al., 1999; Kenny & Rees, 1994, 1996; Kenny et al., 1998; Rees, 1987; Van Moorsel, 1993; Van Moorsel & Waardenburg, 1991). Such comparisons are essential in establishing whether responses to aggregate extraction differ from prevailing natural community dynamics, and when (or if) the disturbed habitat has recovered.

The purpose of this study was to investigate whether different historical levels of dredging intensity affected the subsequent rate of benthic re-colonisation at a marine aggregate extraction site, after the event of cessation. The study is part of an ongoing field-assessment programme designed to enhance the understanding of processes leading to physical and biological recovery of the seabed following dredging, thereby aiding the identification of practices to minimise environmental harm at licensed sites, and to promote rehabilitation on cessation. The outcome of the survey work will allow questions about the present status of dredge sites to be answered directly, rather than by inference.

2. Materials and methods

2.1. Study site

The study site (designated 'Area 222') is located approximately 20 miles east of Felixstowe off the southeast coast of England (Fig. 1) in water depths between 27 and 35 m Lowest Astronomical Tide (LAT). The tidal ellipse in the region is rectilinear and is aligned in a NNE–SSW direction. The local tidal ellipse is thought to be modified slightly by an adjacent deeper channel that encroaches into the northern edge of the site (J. Rees, CEFAS, personal communication). Maximum spring tidal current velocities reach $1.5\,\mathrm{m\,s^{-1}}$ and the CEFAS numerical sediment transport model, tailored

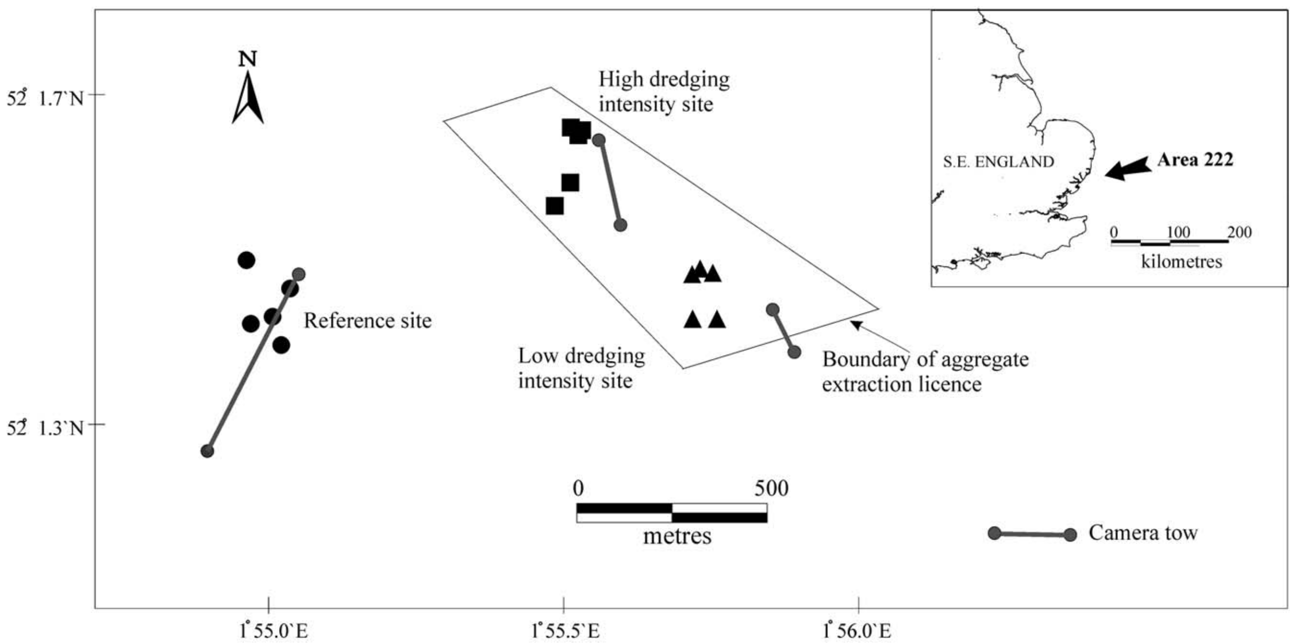

Fig. 1. Map showing the location of Area 222 extraction licence and sample locations in the Southern North Sea.

for studies at aggregate extraction sites, suggests a NNE near-bed residual tidal direction in the vicinity of Area 222. Thus, a similar net bed sediment transport direction can be inferred at Area 222. The surface and sub-bottom geology of the area comprises a basal unit of the London Clay Formation, which has been subsequently overlain by sand and gravel deposits (ARC Marine Ltd, 1997).

This site was first licensed for sand and gravel extraction on 16 December 1971. Historical records indicate that the annual rate of extraction peaked in 1974 at 872,662 t and that extraction was maintained at a somewhat lower level, still in excess of 100,000 t per annum, until 1995 (Fig. 2). The site was last dredged in 1996, when approximately 12,000 t was removed. The dredging company finally relinquished the licence on 31 December 1997 (Crown Estate records). At this site, the sand:gravel ratio in the cargo was adjusted by screening, excess sand being discharged overboard at the site of dredging. By this screening process, the dredging company could realise an optimum mix of sand and gravel for customer requirements.

2.2. Sampling design

Area 222 was not dredged in the 4 years prior to sampling. Therefore, historical information on the location and intensity of dredging was used to direct sampling. Since 1993, every vessel dredging on a Crown Estate's licence in the UK has been fitted with an Electronic Monitoring System (EMS), which consists of a PC, electronically linked to the vessels navigation system and one or more dredging status indicators. This automatically records the date, time and position of all dredging activities, every 30 s, to disk. Many of the dredgers operating in UK waters are fitted with Differential Global Positioning Systems, which allow the EMS to operate with an accuracy of ± 10 m.

EMS information was interrogated in order to locate areas of the seabed within Area 222, which had been subjected to different levels of dredging intensity (Fig. 3). Five replicate samples were collected from areas representing two different levels: (1) >10 h of dredging (high) within a 100 m by 100 m block during 1995; and (2) <1 h of dredging (low) within a 100 m by 100 m block during 1995. The dredging history of each sampling location is presented in Table 1. No records exist on the level of the dredging intensity that these locations were subjected to prior to the introduction of the EMS in 1993. Five replicate samples were also collected from a nearby reference location, which was considered to be representative of the wider environment surrounding this historic extraction licence. Site selection was aided by the use of sidescan sonar and video images of the seabed (see Section 2.4 for methodology). The criteria adopted for selecting a reference site followed those given in CSTT guidelines (1997) and Boyd (2002).

With this design, the area of 'high' dredging intensity represents conditions following the repeated removal of commercial aggregate from most of the total surface area of a 100 m by 100 m block, many times over the course of 1 year. This assumes that a dredger, typically a trailer suction hopper dredger, moves slowly over the seabed at a speed of 2 kt and creates a dredge track approximately 2.5 m wide (Kenny & Rees, 1994; Millner et al., 1977; Van Moorsel & Waardenberg, 1990). It also assumes that the dredger works systematically across an area. In practice, particular deposits will be more frequently targeted by the dredging industry and therefore,

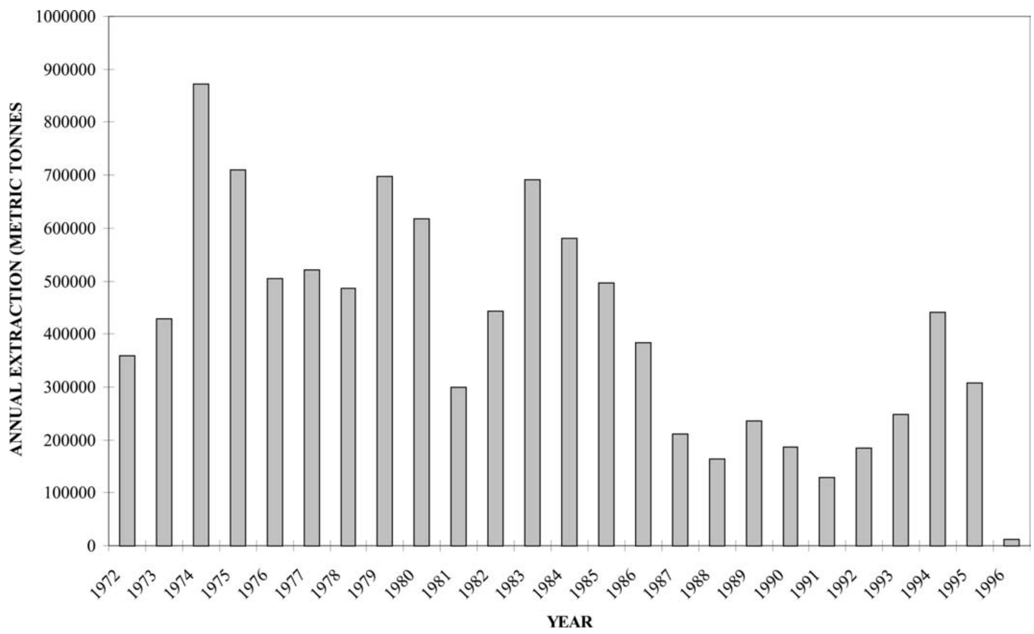

Fig. 2. Annual extraction of marine aggregate extracted from Area 222 since 1972.

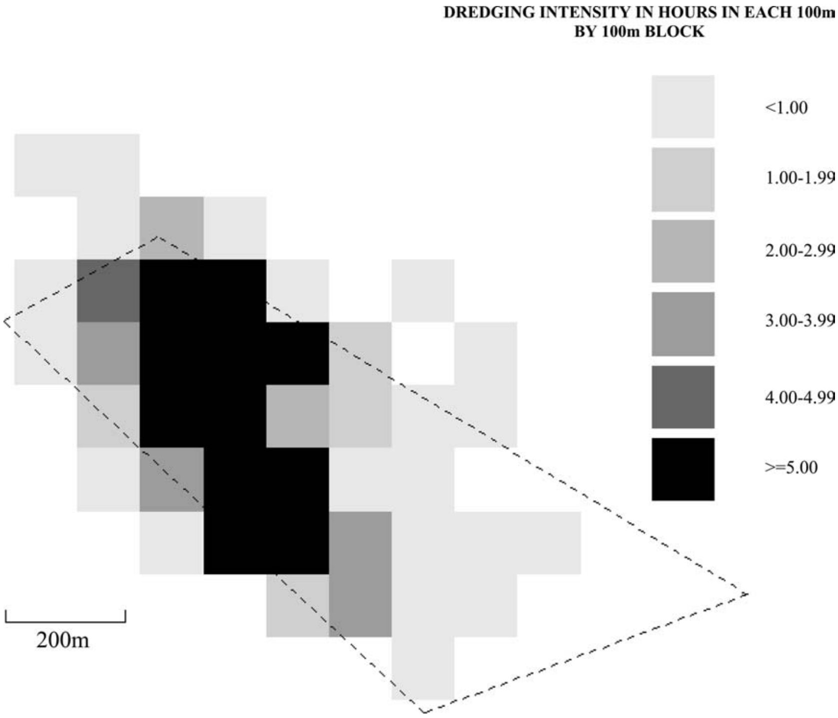

Fig. 3. Block analysis of data extracted from the EMS for hours of recorded dredging intensity in 1995.

under 'high' dredging intensity, some areas of the seabed may have been dredged in the past on a regular basis, whereas other areas of the seabed may only have been dredged once or twice a year. In contrast, the area of 'low' dredging intensity represents conditions after the removal of up to about 90% of the total surface area in a similar 100 m by 100 m block in a single year. However, some locations within this treatment may have only experienced limited exposure to the direct effects of extraction, allowing survival of some species and recolonisation by others while extraction was ongoing. In addition, the 'low' dredging intensity samples were collected approximately 400 m to the south of the area of 'high' dredging intensity (see Fig. 3). Direct studies on sediment settlement suggest that sand is deposited at distances up to 300–600 m down current from a dredger, with the possibility of plume effects extending significantly beyond this (Hitchcock & Drucker, 1996). Therefore, the area of 'low' intensity was potentially subjected to any indirect effects (e.g. transport of unconsolidated sediments) associated with the nearby more intensive dredging activity.

Table 1
Maximum number of hours of recorded dredging at each sample location, within a 100 m by 100 m block, over time (data extracted from the EMS (Crown Estate data))

Sample	1993	1994	1995	1996
High (A)	10.0	13.0	20.0	1.5
High (B)	2.0	17.5	12.5	<1.0
High (C)	2.0	17.5	12.5	<1.0
High (D)	9.0	32.5	17.5	<1.0
High (E)	15.0	40.0	17.5	<1.0
Low (A)	2.0	3.0	<1.0	<1.0
Low (B)	<1.0	2.0	<1.0	<1.0
Low (C)	3.0	3.0	<1.0	<1.0
Low (D)	2.0	3.0	<1.0	<1.0
Low (E)	3.0	3.0	<1.0	<1.0

2.3. Sample collection

Samples for analysis of the macrobenthic fauna and sediment particle size were collected with a 0.1 m^2 Hamon grab from R.V. *Cirolana* in June 2000 (Fig. 1). Five replicate samples were collected from areas of the seabed that had been identified from EMS records in 1995 as being of 'high' and 'low' dredging intensity. Five replicate samples were also collected from a nearby reference site. Values of dredging intensity corresponding to the 10,000 m^2 square block where each sample was collected were then extracted from the EMS for all years of dredging activity (see Table 1). This allowed confirmation that all samples were obtained from areas of the seabed appropriate to their pre-defined treatment group.

Following estimation of sample volume, a 500 ml sub-sample was removed for laboratory particle size analysis. The whole sample was then washed over 5 and 1 mm square mesh sieves to remove the fine sediment. The two resultant fractions (1–5 and >5 mm) were back-washed into separate containers and fixed in 4–6% buffered Formaldehyde solution (diluted in seawater) with the addition of 'Rose Bengal', a vital stain.

2.4. Sidescan sonar and video camera surveys

A sidescan sonar survey using a Datasonics™ SIS 1500 digital chirp sidescan system was conducted in June 2000. This survey was undertaken in order to provide an indication of the spatial distribution of sediments in the wider area encompassing the dredged site and to estimate the likely spatial extent of both direct and indirect effects of dredging. Six survey lines (approximately 3 km long) were run, using E–W and NW–SE line orientations at 400 m line spacing to achieve 100% coverage of the extraction site. The digital data were acquired and post-processed using the TRITON ISIS™ and DELPHWIN™ software packages, producing a geo-referenced, on-screen mosaiced image of the sidescan survey lines. This image was interpreted using ground truth data from grab and underwater video surveys, and the main substrate types and their associated features were delineated and mapped. Further details on the field procedures, post-processing and related data analysis can be found in Boyd (2002).

An underwater video and stills survey was conducted in June 2000 using a Simrad™ video camera and a Benthos DSC™ 4000 digital stills camera mounted within a robust metal frame. This frame was lowered close to the seabed as the vessel drifted with the tide. Video images were recorded automatically onto high-resolution SVHS tapes, and photographic images of features of interest were taken manually and stored directly to a PC on-board the vessel. Deployments of approximately 10 min duration were carried out over the 'high' and 'low' treatment areas, and also at the 'ref' location. This resulted in tow lengths of 230 m ('high'), 130 m ('low') and 520 m ('ref'), respectively.

2.5. Macrofaunal sample processing

Macrofauna samples were processed according to the guidelines given in Boyd (2002). The >5 mm sample fraction was first washed with fresh water over a 1 mm mesh sieve in a fume cupboard, to remove excess Formaldehyde solution, then back-washed onto a plastic sorting tray. Specimens were removed and placed into labelled glass jars containing a preservative of 70% Industrial Methylated Spirits. Specimens were identified, where possible, to species level. The 1–5 mm fraction was first washed over a 1 mm sieve then back-washed into a 10 l bucket. The bucket was filled with fresh water and the sample was then gently stirred in order to separate the animals from the sediment. Once the animals were in suspension, the sample was decanted over a 1 mm mesh sieve. This process was repeated until no more material was recovered. Specimens from this fraction were placed into labelled petri dishes for identification and enumeration. The sediment was then placed on plastic trays and examined under an illuminated magnifier for any remaining animals, such as bivalves not recovered in the decanting process, which were then added to the petri dishes. The blotted wet weight (in milligrams) for each species recorded from replicate samples was also recorded.

2.6. Sediment particle size analysis

The sediment sub-samples from each grab were analysed for their particle size distributions. Samples were first wet-sieved on a 500 μm stainless steel test sieve using a sieve shaker. The <500 μm sediment fraction passing through the sieve, was allowed to settle from suspension in a container for 48 h. The supernatant was then removed using a vacuum pump and the remaining <500 μm sediment fraction was washed into a petri dish, frozen for 12 h and freeze-dried. The total weight of the freeze-dried fraction was recorded. A sub-sample of the <500 μm fraction was then analysed using a laser sizer. The >500 μm fraction was washed from the test sieve into a foil tray and oven dried at ~90 °C for 24 h. It was then dry-sieved on a range of stainless steel test sieves, placed at 0.5 ϕ intervals, down to 1 ϕ (500 μm). The sediment on each sieve was weighed to 0.01 g and the values were recorded. The results from these analyses were combined to give a full particle size distribution for each sample.

2.7. Data analysis

The following univariate measures were calculated: total abundance (A), numbers of species (S), Shannon–Wiener diversity index H' $\log_2$ (Shannon–Weaver, 1949), and Margalef's species richness (d). Ash-free dry weights (AFDW) were calculated using standard conversion factors (Ricciardi & Bourget, 1998; Rumohr, Brey, & Ankar, 1987).

Differences between each level of dredging intensity were determined through calculation of Least Significant Intervals (LSIs). This method assumes that where the means do not overlap, there is a statistically significant difference at the 95% probability level (Andrews, Snee, & Sarner, 1980). The inter-relationships between the dredging history at each sample location over different years, sedimentary variables and a range of univariate indices of biological structure were examined using Pearson product moment correlation co-efficients.

Non-metric multidimensional ordinations of the inter-sample relationships (Kruskal & Wish, 1978) were produced to view the similarity of samples in terms of their species composition. The ordination was based on a lower triangular similarity matrix of fourth root-transformed abundance data using the Bray–Curtis similarity co-efficient (Bray & Curtis, 1957). The transformation reduces the effect of the dominant taxa and hence gives more weight to rare taxa (Clarke & Green, 1988). To test the null hypothesis H_o that there were no significant differences in community composition between samples collected from areas subjected to differing levels of dredging intensity, an analysis of similarities (ANOSIM) procedure was undertaken (Clarke, 1993). The contribution of individual species to observed differences between sample groups was established by means of similarity percentage analyses (SIMPER) on the species sample matrices (Clarke, 1993).

To examine which factors are important in accounting for the distribution of faunal assemblages, information on the dredging history and sediment variables were compared singly or in combination with the ranked dissimilarity matrix of faunal abundance data using the BIO-ENV routine (see Clarke & Ainsworth, 1993). The environmental variables examined were: % silt and clay (<0.063 mm), % sand (0.063–2 mm), % gravel (>2 mm), water depth and the hours of recorded dredging for 1993–1996 derived from the EMS. Sub-sets of environmental variables that were found to best explain the biological variability were then identified by the highest correlation co-efficients (ρ_w). Additional variables, when added to the optimum combination, would be expected to impair the match and hence would have a lower rank correlation. Pearson product moment correlation analyses were performed to establish whether any of the environmental variables were collinear ($r > 0.95$) prior to their inclusion in the BIO-ENV routine. All multivariate analyses were performed using the software package PRIMER, developed at the Plymouth Marine Laboratory (Clarke & Warwick, 1994).

3. Results

3.1. Dredging history of sites

The maximum number of hours of recorded dredging at each sample location for 1992–1996 inclusive is given in Table 1. In 1996, the year that the licensed site was last dredged, the total number of hours of recorded dredging was minimal. The highest value recorded being 1.5 h of dredging in a 100 m by 100 m area. For the years 1993–1995, values of recorded dredging were significantly higher ($p < 0.01$) in the area selected a priori as the area representing high ('high') dredging intensity compared with the area of low ('low') intensity. The highest value of recorded dredging was 40 h of dredging within a 100 m by 100 m area. This was registered by the EMS in 1994, from the 10,000 m^2 block within which sample replicate 'high' (E) was collected. Although this does not necessarily indicate that the entire block was dredged, 40 h of dredging equates to the repeated removal of commercial aggregate from the total surface area of a block of similar size, approximately 37 times over the course of 1 year (see Section 2).

3.2. Water depth

Water depth varied little between sampling locations (range, 27–35 m; LAT). The deeper water was found in the area of 'high' dredging intensity with water depths up to 35 m being recorded. The area of 'low' dredging intensity was found in somewhat shallower water, with maximum depths of 29 m being recorded. Water depths at the reference site were between 30 and 32 m.

3.3. Sediment particle size

Particle size indices from sediments of all treatment groups are presented in Table 2. In general, the sediment from the area exposed to the highest dredging intensity ('high') contained proportionally more sand and less silt/clay than other sampled sediments ('low' and 'ref') (see Table 2). All sampled sediments from the area of 'high' dredging intensity contained less than 2% silt/clay (dry weight). The proportion of sand to gravel from the area of 'high' dredging intensity varied, ranging from 44 to 93% sand (Table 2). In addition, sediments from this treatment group were generally better sorted than other sampled sediments, with sorting co-efficients between 1.00 and 2.48. The lower sorting co-efficients were associated with predominantly sandy samples, and the higher values were a result of the presence of a gravel component.

Sediment samples collected from the area of 'low' dredging intensity were similar to the more gravelly replicates (C–E) from the area of 'high' dredging intensity. Substrates at the reference site were more mixed and variable than those at the dredged sites (Fig. 4). In terms of the % gravel, sediments from the reference site were similar to those of the area of 'low' dredging intensity and several of the replicates (B–D) from the area of 'high' dredging intensity. However, the sand fraction

Table 2
Sediment characteristics from each sample location

Sample location	Mean (mm)	Sorting	Gravel (%)	Sand (%)	Silt and clay (%)
High (A)	0.94	1.00	14.02	85.83	0.15
High (B)	0.64	0.90	6.89	93.11	0.00
High (C)	2.14	2.72	45.28	53.13	1.60
High (D)	2.57	2.52	55.47	44.02	0.51
High (E)	2.53	2.48	52.54	47.00	0.45
Low (A)	1.84	3.13	51.32	43.84	4.84
Low (B)	2.46	2.94	59.03	37.29	3.68
Low (C)	1.79	3.13	51.77	42.80	5.43
Low (D)	1.79	3.20	52.62	41.17	6.21
Low (E)	1.03	3.68	43.99	42.44	13.57
Ref (A)	3.52	3.45	65.77	25.55	8.68
Ref (B)	0.16	4.35	29.94	19.87	50.19
Ref (C)	0.32	4.91	36.74	16.68	46.58
Ref (D)	0.58	4.84	46.42	15.57	38.01
Ref (E)	2.21	3.82	50.47	38.02	11.51

present within the dredged sites was replaced by a silt/clay component at the reference site.

3.4. Sidescan sonar surveys

Predominantly sandy substrates were found within the northern part of the extraction site. These sands have been mobilised to form ripple features. Suction hopper trailer dredge tracks are evident from sidescan sonar records, in the northern part of the extraction site (Figs. 5 and 6). Weathered dredge tracks were also visible from images collected from the south of the extraction site (Figs. 5 and 6). Another area of seabed disturbance, which was characteristic of dredging activity, was located to the north of the extraction site. In this area, the seabed has an uneven profile and appears on the sidescan record as a series of inter-connected pits. There were also a number of sand waves running at right angles to the tidal axis, to the north of Area 222. The presence of these features may be the result of deposition and subsequent entrainment of screened sands produced during the dredging activity within and adjacent to Area 222. Furthermore, immediately to the west of the extraction site, tidally aligned transport features indicate that some bed sediment transport has occurred in this area. The sidescan sonar data indicate that the sub-strata surrounding Area 222, which have not been directly or indirectly affected by historic dredging activity are similar, being composed of a mixture of sands, gravels and occasional outcrops of clay.

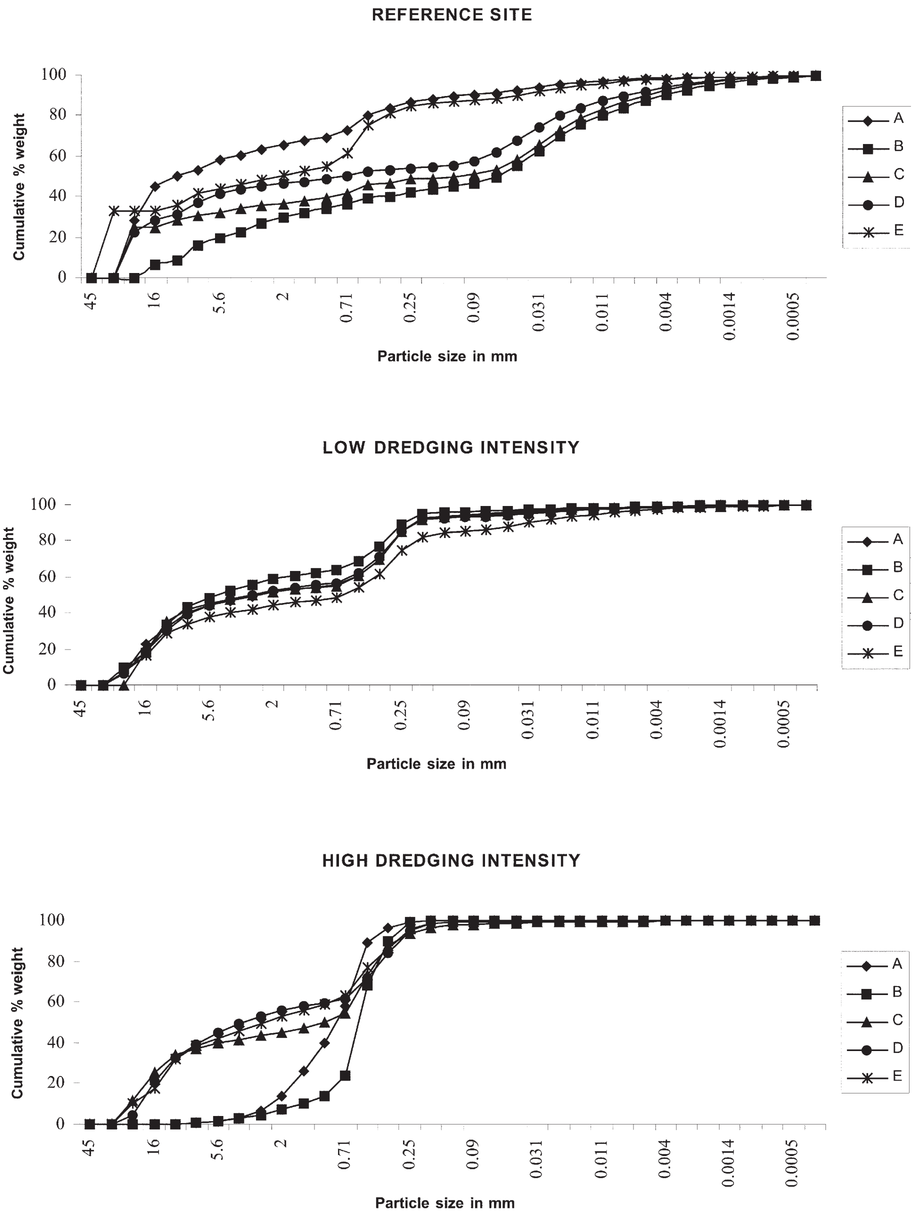

Fig. 4. Sediment particle size distributions determined from replicate samples taken from each level of dredging intensity.

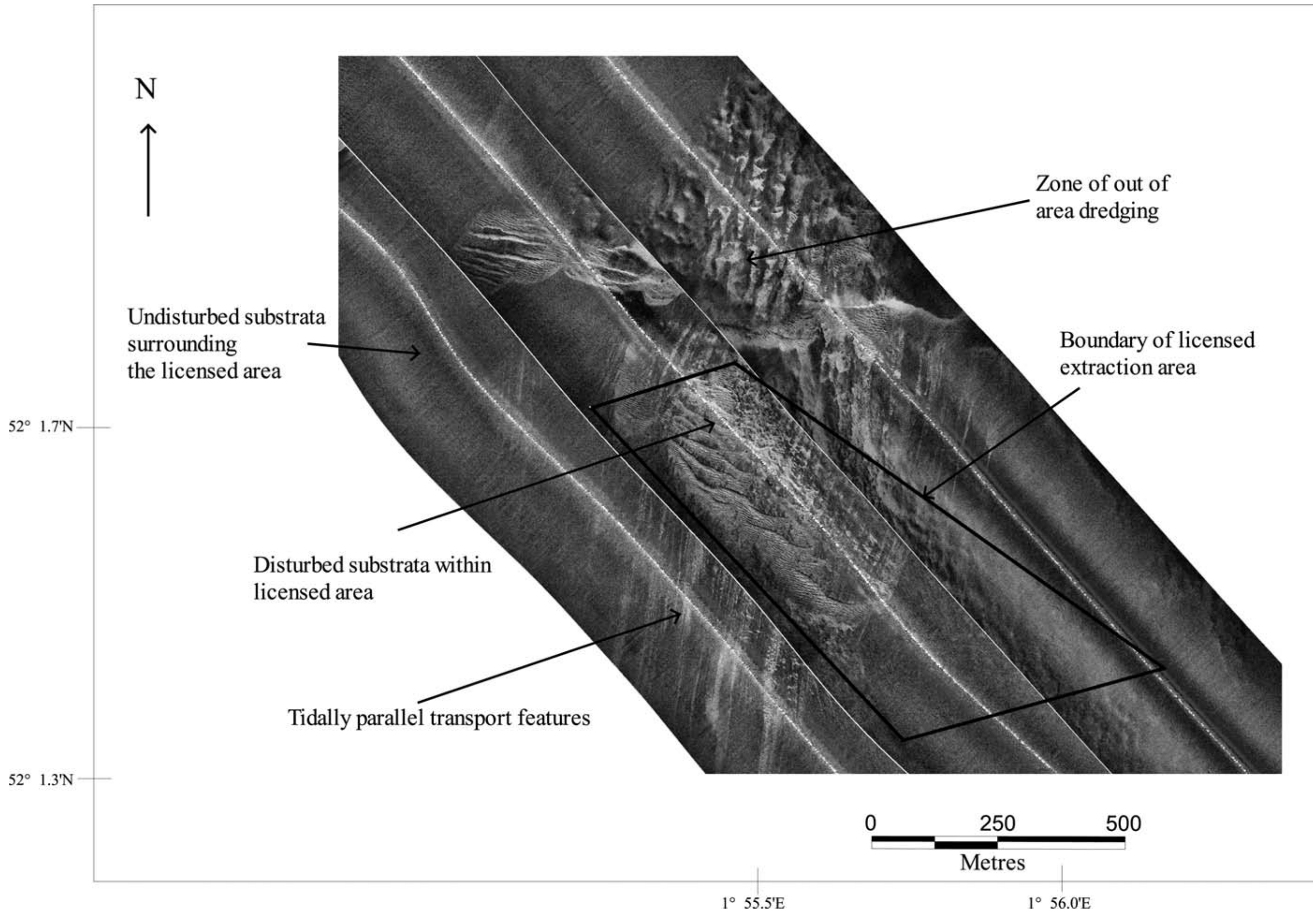

Fig. 5. Sidescan sonar mosaic showing the distribution of substrate types within and surrounding Area 222.

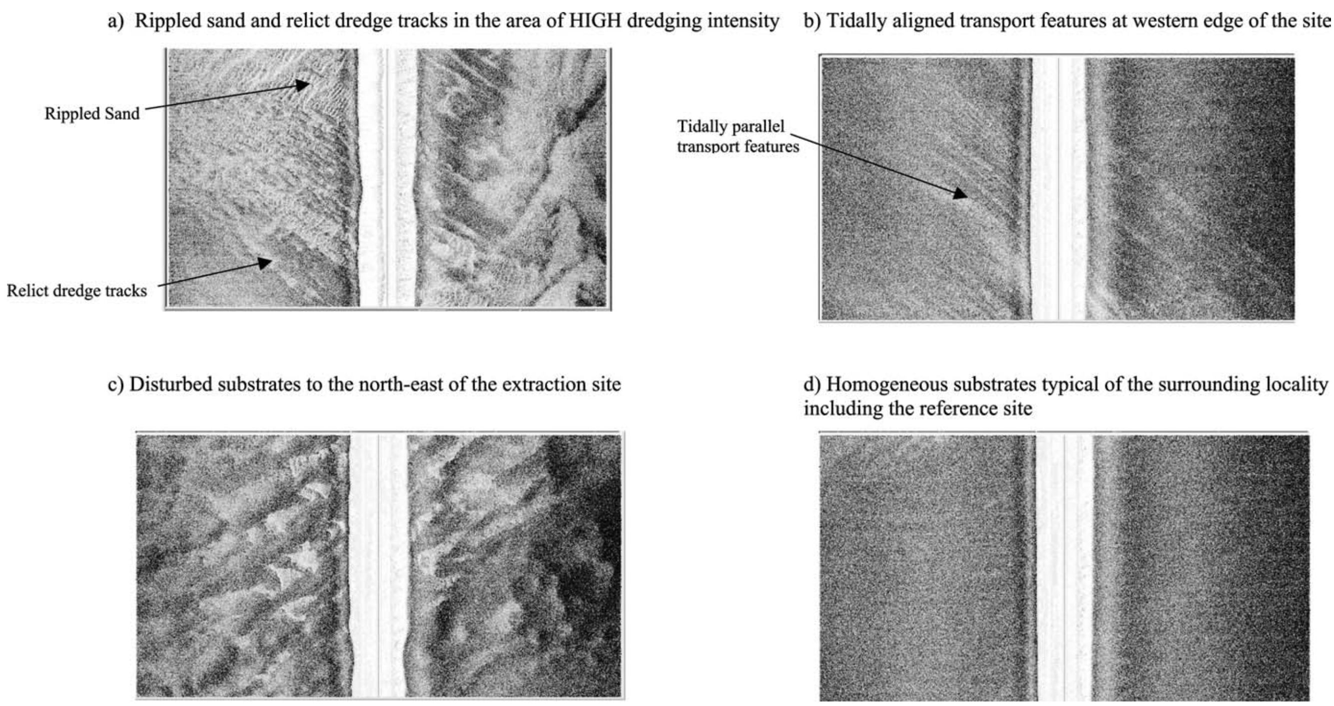

Fig. 6. (a–d) Examples of sidescan sonar records from the sidescan sonar survey. Each image represents a swathe of approximately 400 m.

3.5. Underwater TV survey

Underwater TV images suggest that the bed transport was in the direction of the prevailing tidal flow. Sandy substrates were present in the area of 'high' dredging intensity (Fig. 7) and these were occasionally interspersed with clay outcrops. Towards the northeastern edge of the extraction site, gravels overlain with occasional thin veneers of sand were also visible. Where present, the sand had been mobilised to form 5 m high sand waves and smaller ripple features. In contrast, images collected from the area of 'low' dredging intensity, and from the reference ('ref') site, indicate that here the sub-strata are composed of coarse gravels (Fig. 7).

3.6. Biological observations

A total of 183 taxa were recorded from 15 samples. Of these, 67 were single occurrences. Numerical dominants included the polychaetes *Harmothoe* spp., *Pholoe inornata*, *Glycera lapidum*, *Sphaerosyllis taylori*, *Lumbrineris gracilis*, *Polydora caulleryi*, *Sabellaria spinulosa*, *Pomatoceros triqueter*, the pea crab *Pisidia longicornis*, the bivalve *Mysella bidentata* and the brittle star *Amphipholis squamata*.

A one-way ANOVA detected highly significant differences between levels of dredging intensity for all univariate measures except the values of biomass (AFDW) (Table 3). Values of abundance, number of species, species richness, diversity and evenness are significantly ($p < 0.05$) depressed in the area previously exposed to the highest level of dredging intensity ('high') compared with measures derived from all other samples ('low' and 'ref'; Fig. 8). Macrofauna were present at lowest densities, $260\,m^{-2} \pm 175$ in the area of highest dredging intensity ('high'), at intermediate densities ($1150\,m^{-2} \pm 376$) from the area of low intensity ('low') and at highest densities ($2120\,m^{-2} \pm 371$) from the reference ('ref') site. The average number of non-colonial species recorded from the 'high' sample group was 13 (± 6.94), which was significantly lower than the 'low' sample group (45 ± 8.75) and the reference site (51 ± 6.46). Except abundance, all other univariate measures calculated from reference samples ('ref') were not significantly different ($p > 0.05$) from samples collected from the lowest dredging intensity ('low'). The mean biomass (AFDW) was $0.133\,g\,m^{-2}$ (± 0.07) for sediments from the area of highest dredging intensity, compared with $4.65\,g\,m^{-2}$ (± 1.73) from the 'low' and $5.7\,g\,m^{-2}$ (± 1.97) from the reference samples. This reflects the absence of a range of macrofaunal species from the 'high'

Fig. 7. (a–d) Underwater photographic images taken from within and outside Area 222. Each image represents an area of seabed of approximately 1.5 m by 1.0 m.

Table 3
F-ratios and significance levels (from $F_{2,12}$) from one-way ANOVA tests for differences in various univariate measures

Variable	F-ratio	p
Abundance	41.80	<0.01
Number of species	37.38	<0.01
Diversity (H')	14.26	<0.01
Richness (d)	32.26	<0.01
Evenness (J')	9.75	<0.01
Biomass (AFDW)	3.85	0.05

intensity samples including the pea crab *Pisidia longicornis*, the errant polychaetes *Lumbrineris gracilis* and *Marphysa bellii* and *M. sanguinea*, the sedentary polychaetes *Pomatoceros triqueter*, *Sabellaria spinulosa* and the brittle star *Amphipholis squamata*.

The MDS ordination (Fig. 9a) indicates that the samples collected from the area of most intensive dredging ('high') are clearly separated from the area of less intensive dredging ('low') and the reference samples ('ref'). The samples are arranged in sequence across the x-axis with the reference site samples arranged on the right-hand side of the plot and all the high dredging intensity samples ('high') on the left-hand side. Low dredging intensity samples ('low') occupy an intermediate position. The 'low' and 'ref' samples are also separated across the y-axis of the two-dimensional ordination.

Results of SIMPER and ANOSIM analyses also reveal that the differences between 'high' and 'ref' samples are more pronounced than between 'low' and 'ref' samples (Table 4). In addition, ANOSIM analyses confirm that all of the assemblages are significantly different at $p < 0.01$ (Table 4). Furthermore, the individual sample replicates from the area of intensive dredging ('high') are widely separated on the MDS plot, indicating that they are biologically dissimilar (average similarity = 32%).

Differences between previously dredged sediments ('high' and 'low') and the reference location ('ref') are due to the reduced abundance of a range of macrofaunal species characterising nearby sediments ('ref') including *Pomatoceros triqueter*, *Pisidia longicornis*, *Sabellaria spinulosa*, *Amphipholis squamata*, *Polydora flava*, and *Harmothoe* spp. (Table 5). Amongst adults, only the polychaetes *Hesionura elongata*, *Spio filicornis* and *Spiophanes bombyx*, which are typically associated with sandy sediments, were more abundant from the high dredging intensity samples than other sampled locations. Newly settled juveniles of the polychaetes *Nephtys* spp., *Prionospio* spp. and Sabellidae were also more frequent in the high intensity samples.

3.7. Correlations between variables

Further insights into the re-colonisation of the macrofauna following marine aggregate extraction may be

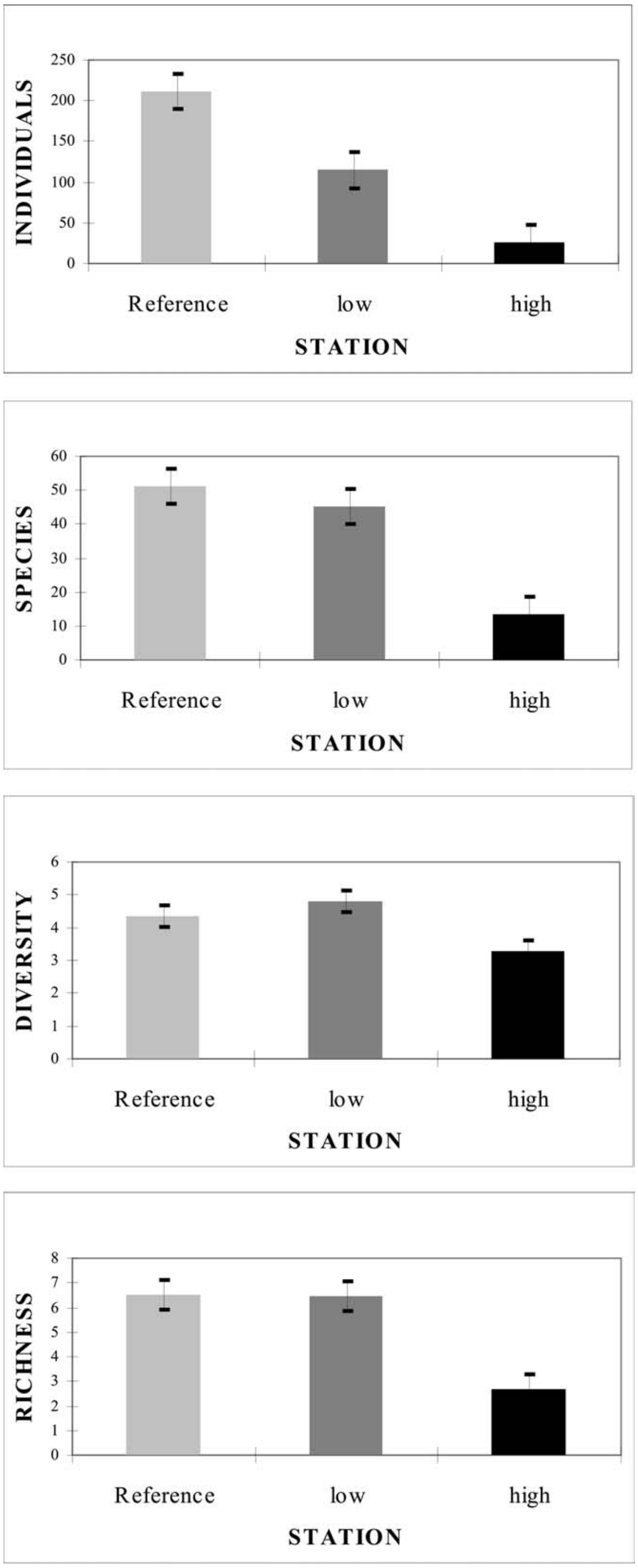

Fig. 8. Means and 95% LSIs for univariate measures of community structure 4 years after the cessation of dredging at high and low levels of dredging intensity.

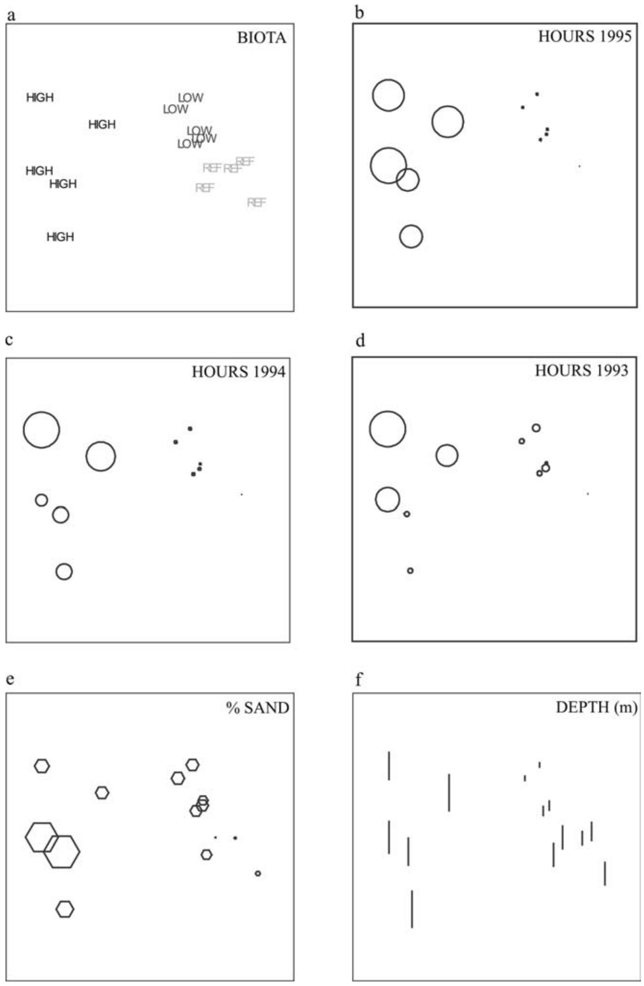

Fig. 9. (a) MDS of Bray–Curtis similarities from double square-root-transformed species abundance data at the three stations; (b–d) the same MDS, but with superimposed circles proportional in diameter to hours of recorded dredging derived from EMS for each year; (e, f) the same MDS, but with superimposed symbols representing values of % sand and depth of water (Stress = 0.09).

Table 4
Dissimilarities (%) between reference samples and other sample groups ('high' and 'low') based on fourth root-transformed species abundance data

	High	Low
Low	82*	
Ref	88*	63*

* Significant differences at $p < 0.01$, based on ANOSIM test.

obtained from an examination of the inter-relationships between the assemblage structure and measured environmental variables including sedimentary factors, water depth and the hours of recorded dredging (Tables 6 and 7).

There was a negative relationship between the hours of recorded dredging and the primary biological measures, especially in the case of hours of dredging in 1995 vs. numbers ($r = -0.8951$, $p < 0.0001$) and densities ($r = -0.832$, $p < 0.0001$) of taxa (Table 7). The degree of sediment sorting and the % sand were the main sedimentological variables explaining the macrobenthic assemblage structure, although % silt and clay was also significantly related. In addition, there are statistically significant relationships between biological measures, particularly diversity, and water depth. Mean particle size and % gravel were not significantly related to any of the biological measures of community structure.

In a standard product moment correlation analysis of all of the environmental variables, hours of dredging (1993–1996) and sediment sorting co-efficient were significantly related ($p < 0.05$; Table 6). With the exception of hours of recorded dredging in 1996 and 1994, dredging intensity is strongly positively correlated between the years. However, only the degree of sediment sorting and % sand were highly correlated ($r > 0.94$). Thus, all the variables were used in the BIO-ENV analysis and the highest few co-efficients at each level of complexity have been tabulated, allowing the extent of improvement

Table 5
The average abundance of the top 15 ranked species 0.1 m^{-2} contributing to the dissimilarity between sample groups derived from SIMPER analyses of fourth root-transformed data (species are ordered in decreasing contribution to the average dissimilarity)

Species	Low	High	Species	Ref	High	Species	Ref	Low
Exogone verugera	5.40	–	*Pomatoceros triqueter*	48.60	–	*Pisidia longicornis*	22.20	0.60
Mysella bidentata	11.20	–	*Pisidia longicornis*	22.20	–	*Pomatoceros triqueter*	48.60	3.00
Notomastus latericeus	4.80	–	*Schistomerigos rudolphi*	16.40	0.4	*Harmothoe* spp. indent	6.00	–
Lumbrineris gracilis	11.00	0.40	*Amphipholis squamata*	7.40	–	*Polydora flava*	2.80	–
Ascidiacea spp. juveniles	2.60	–	*Polydora caulleryi*	6.60	–	*Corophium sextonae*	3.20	–
Spio filicornis	2.80	2.00	*Harmothoe* spp. indent	6.00	–	*Sabellaria spinulosa*	5.20	0.20
Pomatoceros triqueter	3.00	–	*Pholoe inornata*	4.40	–	*Notomastus latericeus*	0.40	4.80
Opiuroidea spp. juveniles	1.80	–	*Polydora flava*	2.80	–	*Heteranomia squamula*	2.20	–
Echinocyamus pusillus	1.80	0.20	*Corophium sextonae*	3.20	–	*Mysella bidentata*	1.20	11.20
Paradoneis lyra	4.20	0.20	*Heteranomia squamula*	2.20	–	Anomiidae spp. indent	1.80	–
Hesionura elongata	–	2.20	Anomiidae spp. indent	1.80	–	*Amphipholis squamata*	7.40	2.00
Exogone naidina	2.00	–	Amphiuridae spp. juveniles	3.40	–	*Spiophanes bombyx*	0.40	2.80
Lanice conchilega	1.60	–	*Lanice conchilega*	2.20	–	*Dendrodoa grossularia*	5.20	–
Sphaerosyllis taylori	9.80	2.40	*Sabellaria spinulosa*	5.20	1.20	*Janira maculosa*	0.60	1.80
Laonice bahusiensis	1.00	–	*Hesionura elongata*	–	2.20	Amphiuridae spp. juveniles	1.40	–

Table 6
Pair-wise correlations between environmental variables

	Dredging in 1996 (h)	Dredging in 1995 (h)	Dredging in 1994 (h)	Dredging in 1993 (h)	Gravel (%)	Sand (%)	Silt and clay (%)	Sediment sorting	Mean particle size (mm)
Dredging in 1995 (h)	**0.6172**								
Dredging in 1994 (h)	0.4977	**0.8774**							
Dredging in 1993 (h)	**0.5952**	**0.8262**	**0.8576**						
Gravel (%)	−0.1977	−0.3806	−0.0768	−0.0762					
Sand (%)	**0.7431**	**0.6999**	0.4514	0.4364	−0.6290				
Silt and clay (%)	**−0.7624**	**−0.5362**	−0.5037	−0.4852	−0.1336	**−0.6864**			
Sediment sorting	**−0.7846**	**−0.7716**	**−0.5747**	**−0.5562**	0.4426	**−0.9477**	**0.7942**		
Mean particle size (mm)	0.1431	0.1195	0.2861	0.2298	**0.7463**	−0.0416	−0.6452	−0.1930	
Water Depth (m)	0.0208	0.7385	**0.6243**	0.4249	−0.3129	0.3506	−0.1542	−0.3583	0.0734

Values in bold type indicate significant correlation ($p < 0.05$).

or deterioration in the match between the biota and environmental variables to be traced as further variables are added (see Table 8). The best fit between the macrofauna community patterns and a single environmental variable is achieved with hours of recorded dredging intensity in 1995 ($\rho_w = 0.826$). Various other combinations of variables with hours of recorded dredging in 1995 give high correlations ($r > 0.8$). The fit is only marginally improved by the addition of other combinations of variables, the highest correlation being 0.829 with a combination of six variables: hours of recording dredging in each year from 1996 to 1993, % sand and water depth.

Fig. 9b–d provides a visual expression of the relationships between macrofaunal data and hours of recorded dredging, with the samples collected from the area of highest dredging intensity ('high') clearly separated from the rest. Samples from the other two groups ('low' and 'ref') are arranged from left to right according to dredging intensity (Fig. 9b–d). The relationship between macrofaunal community structure and both % sand and water depth is less clear (Fig. 9e, f).

4. Discussion

4.1. Survey design

Ideally, reference site(s) in studies such as this, should be identical in all respects to the dredged locations, save for the impact of marine aggregate extraction. However, after dredging has taken place for many years, the benthos (and sediments) may have been structurally altered as a consequence of dredging (Bonsdorff, 1983; Desprez, 1995; Dickson & Lee, 1973; Essink, 1997; Jewett et al., 1999; Jones, 1981; Jones & Candy, 1981; Kaplan, Welker, Kraus, & McCourt, 1975; Kenny & Rees, 1994, 1996; Kenny et al., 1998; Shelton & Rolfe, 1972; Van der Veer, Bergman, Beukema, 1985). In this situation, it is difficult to reach a judgement as to whether a suitably located reference station, in the near vicinity of the dredged site, is representative of the likely pre-dredged status. The reference site in this study was chosen by taking account of both sidescan sonar and video images of the seabed, in an attempt to select a site representative of the wider environment. This resulted in the selection of a reference site varying in sedimentary characteristics with respect to the dredged sediments.

4.2. Physical consequences

The area subject to the highest dredging intensity had a greater proportion of sand than the reference site. There are two possible explanations for this: (1) that sand is naturally present coincident with the area of high dredging intensity, existing as very localised deposits; or (2) that sand has proportionally increased in the area as a consequence of dredging; or (3) a combination of (1) and (2). Without further evidence, it is not possible to definitively conclude whether differences, in sediment granulometry, are natural in origin, or are as a result of dredging activity.

Table 7
Pair-wise correlations between environmental variables and univariate measures of community structure ($0.1\,m^{-2}$)

	Dredging in 1996 (h)	Dredging in 1995 (h)	Dredging in 1994 (h)	Dredging in 1993 (h)	Gravel (%)	Sand (%)	Silt and clay (%)	Sediment sorting	Mean particle size (mm)	Water depth (m)
Total number of species (S)	**−0.6307**	**−0.8951**	**−0.7973**	**−0.7024**	0.3156	**−0.7379**	**0.6455**	**0.7658**	−0.2059	**−0.6876**
Total abundance (A)	**−0.8321**	**−0.8029**	**−0.7494**	**−0.6927**	0.2310	**−0.7376**	**0.7242**	**0.7682**	−0.1642	−0.3718
Species richness (d)	**−0.5370**	**−0.8874**	**−0.7566**	**−0.6574**	0.3722	**−0.7240**	**0.5748**	**0.7376**	−0.1705	−0.7657
Diversity (H')	−0.2725	**−0.7545**	**−0.6191**	−0.5012	0.3777	**−0.5278**	0.3195	0.5047	−0.0712	−0.8142

Values in bold type indicate significant correlation ($p < 0.05$).

Table 8
Summary of results from BIO-ENV (combinations of variables giving the highest rank correlations between biotic and abiotic similarity matrices)

Number of variables	Best variable combination	Spearman rank correlation (ρ_w)
1	1995	0.826
1	1994	0.768
1	1993	0.530
2	1995, 1994	0.805
2	1995, 1993	0.795
2	1994, %S	0.785
3	1995, 1994, %S	0.811
3	1995, 1993, %S	0.809
3	1993, %S, depth	0.796
4	1995, 1993, %S, depth	0.817
4	1995, 1994, %S, depth	0.805
4	1995, 1994, 1993, %S	0.801
5	1996, 1995, 1993, %S, depth	0.826
5	1995, 1994, 1993, %S, depth	0.813
5	1996, 1994, 1993, %S, depth	0.812
6	**1996, 1995, 1994, 1993, %S, depth**	**0.829**
6	1995, 1994, 1993, sorting, %S, depth	0.817
6	1996, 1995, 1994, 1993, %G, depth	0.810

Macrofauna data are fourth root-transformed. The highest correlation is shown in bold. Lower correlations are omitted from the table.
1996: Hours of recorded dredging in 1996.
1995: Hours of recorded dredging in 1995.
1994: Hours of recorded dredging in 1994.
1993: Hours of recorded dredging in 1993.
%S, % sand; %G, % gravel; depth, water depth (m).

4.3. Biological consequences

Both the univariate and multivariate measures of community structure indicate that there are significant differences in the macrofaunal assemblages between areas exposed to different dredging intensities. The area that was previously dredged at a high intensity had a reduced complement of species and, substantially, lower densities of individual macrofaunal organisms than other sampled sediments. Replicate samples from this location were also very dissimilar in terms of the species composition, which is often symptomatic of disturbed habitats (see, for example, Clarke & Warwick, 1994; Kenny & Rees, 1994). Univariate and multivariate analyses also indicate a strong relationship between macrofaunal community structure and dredging intensity at Area 222. In addition, the BIO-ENV and MDS analyses clearly demonstrate that, of the environmental variables measured, dredging intensity, or some other associated variable, is the single-most important factor in determining the macrofaunal community composition at this site. Furthermore, correlation analyses suggest that the predominant influence on the macrofaunal community is the level of dredging that took place in 1995. This was the last year that the licensed site was dredged heavily with a total of 307,418 t of sand and gravel being removed from the site (see Fig. 2). What facet of the dredging activity caused these changes, cannot be answered directly, as dredging intensity was simply used as a measure of the level of dredging disturbance.

Correlation analyses from commercial aggregate extraction licences located east of the Isle of Wight, English Channel (Boyd & Rees, 2003) also implicated dredging intensity as a determinant of macrofaunal community structure, although in this study, aggregate extraction was ongoing. In the present study, commercial marine aggregate extraction had ceased 4 years previously, but effects on the fauna were still apparent. Boyd and Rees (2003) showed that increasing dredging intensity amplifies the proportion of species affected by dredging disturbance, thereby potentially prolonging the time-scale for re-establishment of the benthic community. Other studies have shown that, after an initial increase in the overall abundance and the number of species during the early stages of post-dredging re-colonisation, further progress towards biological 'recovery' is dependent on the habitat attaining a degree of physical stability (Kenny & Rees, 1996; Kenny et al., 1998). Until this occurs, the increased bed-load transport (usually sand) inhibits survivorship and growth of new recruits. Over time, the bed-load transport would be expected to approach the pre-dredged equilibrium, allowing the restoration of pre-dredging levels of abundance and biomass (ICES, 2001; Kenny et al., 1998). In cases, where unconsolidated sediments are regularly disturbed by tidal currents or wave action, it has been suggested that communities may be maintained at an early successional stage (Kenny & Rees, 1994). In the present study, the assemblage from the area of high dredging intensity was largely composed of juveniles suggesting that these individuals are unable to reach adulthood due to the unstable sediments. After repeated dredging, the transport of fines out of the area by waves and tides may not keep pace with the considerable quantities generated during dredging, particularly if elevated levels of intensity are sustained over many years. In such circumstances, it is anticipated that physical 'recovery' of the habitat would be postponed until sediments have either consolidated and/or the burden of sand has moved out of the area. Further, the potential for biological 'recovery' would also be prolonged until such time as physical stability of the habitat is achieved.

4.4. Rates of biological and physical recovery

The estimated time required for re-establishment of the benthic fauna following marine aggregate extraction may vary depending on the nature of the habitat, the scale and duration of disturbance, hydrodynamics and associated bed-load transport processes, the topography of the area and the degree of similarity of the habitat with that which existed prior to dredging (for review see Newell, Seiderer, & Hitchcock, 1998). However, available evidence, largely obtained from experimental studies, suggests that substantial progress towards 'recovery' could be expected within 2 to 3 years of cessation of dredging in sandy gravel habitats exposed to moderate wave exposure and tidal currents (van Dalfsen et al., 2000; Desprez, 2000; Desprez & Duhamel, 1993; de Groot, 1979; Kenny et al., 1998; Newell et al., 1998; Van Moorsel, 1993). Preliminary observations from the present study indicate that the 'recovery' period may be more prolonged, especially for sites dredged repeatedly. The length of time for which dredge tracks or pits remain as recognisable features on the seabed is dependent on the ability of tidal currents or wave action to erode crests or transport fine sediments into them (Dickson & Lee, 1973; McGrorty & Reading, 1984; Millner et al., 1977; Van der Veer et al., 1985). Dredge tracks have been observed to persist between 3 and >7 years (Essink, 1997; Kenny & Rees, 1996; Millner et al., 1977). In a water depth of 25 m, dredge tracks appeared to have been completely eroded within 3 years at an experimental site off North Norfolk, UK. In this case, infill was considered to be the result of sand in transport. In contrast, at a high energy site on the Klaverbank, dredge tracks (of 0.3–0.5 m deep) were found to completely disappear in a gravelly sub-strata at a depth of 38 m within 8 months (Van Moorsel, 1993, 1994; Van Moorsel & Waardenburg, 1991). In contrast, depressions created by static dredging have been reported to persist for several years at a location of Hastings in the Eastern English Channel (Shelton & Rolfe, 1972). Dickson and Lee (1973) concluded that at this location, many years, perhaps amounting to decades, would be required for the dredged seabed to return to its pre-dredging condition. Indeed, the disturbed area to the north east of Area 222, evidently dredged prior to 1993 (i.e. before the introduction of EMS), suggests that such extended periods will be required for the erosion of dredge tracks/pits and for the seabed to revert to its pre-dredged state (see Dickson & Lee, 1973). Further studies at contrasting locations are being undertaken as part of this wider study in order to establish time-scales of physical and biological recovery in the aftermath of dredging.

Acknowledgements

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Small-scale Mapping of Sea-bed Assemblages in the Eastern English Channel Using Sidescan Sonar and Remote Sampling Techniques

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A survey was conducted in the eastern English Channel to investigate the use of sidescan sonar, used in conjunction with traditional biological sampling methods, to map the variety and distribution of benthic biotopes (i.e. sea bed habitats and their associated biological communities). An area of sea-bed, approximately 28 km × 12 km in size, offshore from Shoreham, U.K., was surveyed using a digital sidescan sonar system and a mosaic of the output was produced covering 100% of the survey area. This was used to divide the area into acoustically distinct regions, around which subsequent benthic ground-truth surveys were designed. Benthic communities and sediment types within each of the these regions were sampled using a Hamon grab fitted with a video camera, and using a heavy duty 2-m beam trawl. Further information concerning the sea-bed was obtained through the application of additional video and photographic techniques. Substrates within each acoustic region were generally homogeneous in distribution, and sediment types ranged across the survey area from cobbles and coarse gravel through to muddy sands. Analysis of the faunal data revealed the presence of statistically distinct biological assemblages within most of the acoustic regions, although species similarity between samples collected from within each acoustic area was often low. Using a combination of all the data sets, five discrete biotopes could be identified and mapped across the area. The application of acoustic techniques, used in conjunction with biological sampling techniques, to map the distribution of sea-bed habitats and associated benthic communities is discussed.

Keywords: mapping; sidescan sonar; biotope; habitat; benthic community

Introduction

Efforts to describe and interpret the variability and distribution of benthic fauna over large geographical areas date back to the classic studies of Petersen and co-workers in Danish waters (e.g. Petersen, 1918). In the eastern English Channel the fauna of large sectors of sea-bed are well documented by Holme (1961, 1966) and, on the French side, by Cabioch and co-workers (Cabioch, 1968). Many of these earlier mapping studies, along with most other benthic surveys, have traditionally used grabs and/or dredges to describe the invertebrate fauna of the sea floor. Such techniques provide single, geographically separated points of data across the area of sea-bed under investigation. In order to produce spatial distribution maps of sediments and assemblages from such sources of data it is necessary to interpolate between these data points. However, interpolation has the potential to overlook discrete sea-bed features and/or biological assemblages, which may lie between sampling stations. Recent developments in acoustic technologies may provide a solution to this problem, particularly when attempting to describe the spatial distribution of habitats/assemblages over relatively small areas, and are offering new insights and opportunities to explore and map sea-bed habitats.

The first sidescan sonar systems were developed in the 1940s (Fish & Carr, 1990) allowing crude and rather low resolution images of the sea-bed to be produced. Rapid developments in acoustic electronics in the 1970s and 80s led to major improvements in image quality and, whilst the use of such systems was focused mainly on geological studies of the sea-bed, it was at this stage that the potential use of sidescan sonar for studying benthic ecosystems was recognized (e.g. Warwick & Davies, 1977; Holme & Wilson, 1985; Davoult & Clabaut, 1988). Recent improvements in sidescan sonar systems in the 1990s, as a

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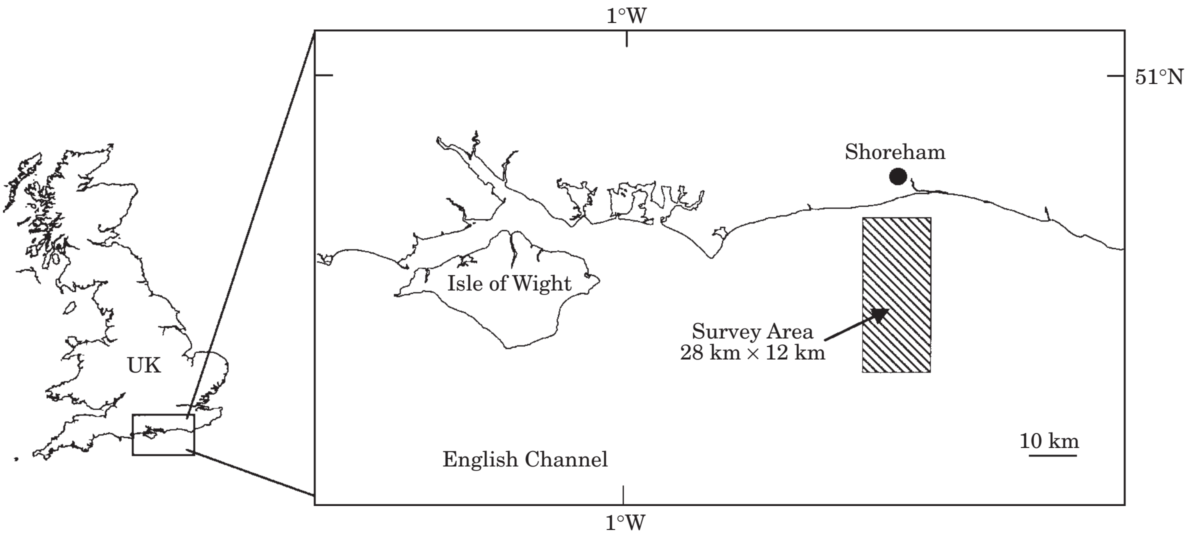

FIGURE 1. Geographical location of the survey area.

result of the increased digital processing power offered by modern computers, have led to very high resolution (almost photographic quality) and affordable systems entering the market place. This development is reflected in the number of recent investigations which have used this technique as a means to infer the biological status of the sea bed (Phillips *et al.*, 1990; Schwinghamer *et al.*, 1996, 1998; Service & Magorrian, 1997; Service, 1998; Tuck *et al.*, 1998; Wildish & Fader, 1998). These developments offer the opportunity for researchers to move away from a process of inference around a matrix of spot samples into the realm of spatially continuous mapping using spot sampling for ground-truthing. For this reason the use of acoustic techniques to assist in mapping the geographical distribution of biotopes (e.g. physical habitats and associated biological communities) can be seen to have many potential advantages, including the prospect of 100% coverage of the sea-bed as resources allow or priorities dictate.

In this paper, the utility of high-resolution sidescan sonar for use in mapping sea-bed assemblages at relatively small scales over coarse substrates in the Eastern English Channel is evaluated. This work formed part of a wider study, funded by the U.K. Ministry of Agriculture, Fisheries and Food, which aimed to investigate methods for mapping sea-bed biotopes in areas of coarse substrates, and establish which factors control the distribution, type and diversity of their biological communities. The relationships between acoustic output, physical habitat and benthic assemblage structure are examined and the potential utility of sidescan sonar in routine sea-bed mapping activities is appraised.

Materials and methods

Acoustic survey

Following preliminary underwater video surveys in May 1999, an area of sea-bed in the English Channel off Shoreham (12 km × 28 km) appeared to offer a suitable site for the study (Figure 1), consisting of a range of homogeneous substrate types. An intensive survey of the area was conducted in July 1999 using a Datasonics digital chirps sidescan sonar with a Triton Isis logging system. Delphmap post-processing software was used to mosaic the imagery and classify texturally different regions. The system was operated on a 400-m swathe range, and survey lines were spaced at 400-m intervals in a north–south orientation in order to ensonify 100% of the survey area. Vessel position was provided by the Veripos Differential Global Positioning system (DGPS) and towed sensor position calculated by vessel heading, towcable layback and towfish depth, all of which were logged in real time by the Isis system.

A drop-camera frame fitted with an under-water video camera and light was deployed at a number of stations across the survey area in order to provide visual ground-truth data to aid interpretation of the sidescan sonar data set. Following the cruise the sidescan mosaic was used to divide the survey area into eight acoustically distinct regions.

Benthic survey

The design of the macrobenthic grab survey was structured around the eight acoustically distinct

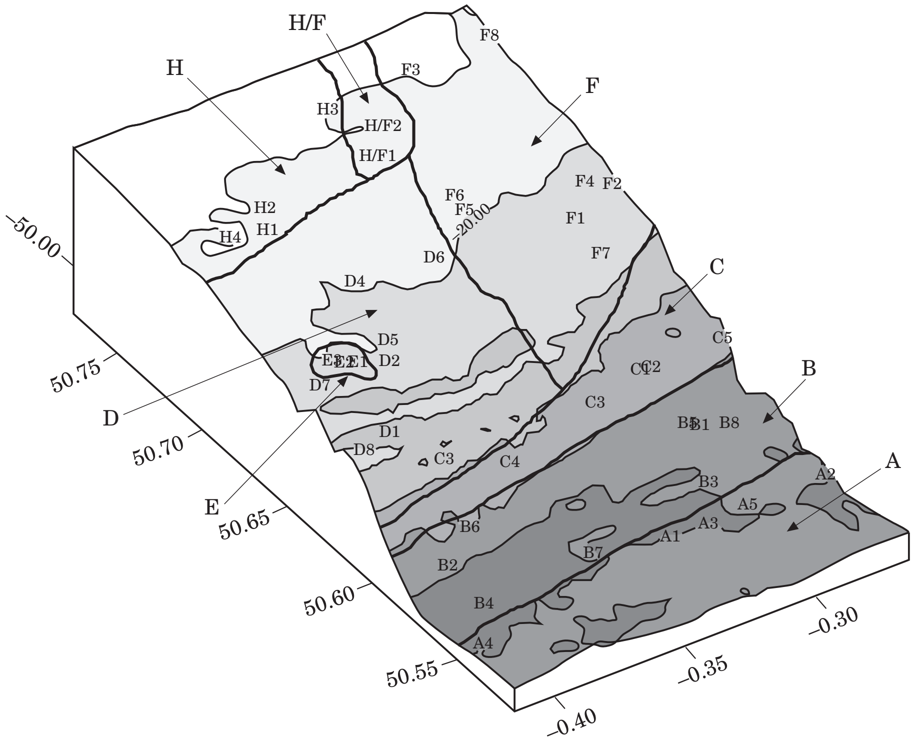

FIGURE 2. Bathymetric plot of the survey area showing the eight acoustically distinct regions (A, B, C, D, E, F, H, H/F) determined from the sidescan sonar data, and locations of the sampling stations.

regions identified from the output of the acoustic survey. Sampling stations were randomly positioned within each acoustic region, and the number of stations within each region was linked to the size of the area (Figure 2). The survey was conducted in August 1999. A total of 43 samples were collected from across the survey area using a 0·1-m^2 Hamon grab fitted with a video camera and light. The camera recorded an image of the sea-bed adjacent to the collection bucket of the grab, thus providing information about the undisturbed surface of the substrate at each sample station. Following estimation of the total sample volume, a 500-ml sub-sample was removed for laboratory particle size analysis. The remaining sample was washed over 5-mm and 1-mm square mesh sieves to remove excess sediment. The retained macrofauna were fixed in 4–6% formaldehyde solution (diluted with seawater) and returned to the laboratory for identification and enumeration.

A beam trawl survey was conducted in July 2000. A modified 2-m beam trawl, with a heavy-duty steel beam, chain mat and a 4-mm knotless mesh liner fitted inside the net (see Jennings *et al*., 1999 for design specifications) was deployed at three randomly selected sampling stations within six of the acoustically distinct regions. The beam trawl was deployed from the stern ramp of RV *Cirolana* using a warp length of three times the water depth. Each tow covered a fixed distance of 120 m across the sea-bed, which was determined using Sextant software linked to the ship's Differential Global Positioning system. The speed of the ship and the time that the gear was on the sea-bed was also recorded. On retrieval of the trawl, an estimate of the sample volume was made. Each sample was washed over a 5-mm square mesh sieve and macrofaunal species were identified and enumerated at sea. Colonial species were recorded as either present or absent. Any specimens which could not be identified at sea were fixed in 4–6% formaldehyde solution and returned to the laboratory for identification.

In the laboratory Hamon grab samples were first washed with fresh water over a 1-mm square mesh sieve in a fume cupboard to remove the excess formaldehyde solution. Samples were then sorted and the specimens placed in jars or petri-dishes containing a preservative mixture of 70% methanol, 10% glycerol and 20% tap-water. Specimens were identified to species level, or as far as possible, using standard taxonomic keys. The number of each species was recorded, and colonial species were recorded as present or absent. For each positive identification a representative specimen was retained in order to establish a reference collection.

The sediment sub-samples from each grab station were analysed for their particle size distributions. Samples were first wet sieved on a 500-micron stainless steel test sieve, using a sieve shaker. The sediment fraction less than 500 microns, along with water from the wet sieving, was allowed to settle in a bucket for 48 h. Excess water was then removed using a vacuum pump and the fraction was washed into a sterile petri dish, frozen for 12 h and freeze dried. The weight of the sediment was also recorded. A sub-sample of the <500 micron freeze dried fraction was then analysed on a laser sizer. The >500 micron fraction was washed from the test sieve into a foil tray and oven dried at ~90 °C for 24 h. It was then dry sieved for 10 min on a range of stainless steel test sieves at half phi intervals, down to 1 phi. The sediment on each sieve was weighed to 0·01 g and the results recorded. The results from these analyses were combined to give the full particle size distribution. The mean, sorting and skewness values were then calculated.

Data processing

Univariate analysis. Total number of individuals (excluding colonial species) and total number of species were calculated from both the Hamon grab and beam trawl surveys as summary measures of benthic assemblages within each acoustic region. Bartlett's test was used to test for homogeneity of variance. A log transformation of the data was required before the variance was homogeneous. The significance of differences between acoustic regions was tested using one-way ANOVA. Fisher's least significant difference (LSD) multiple comparisons procedure was used to determine significant differences in the numbers of species and number of individuals between regions. Univariate analyses were performed using the software package STATGRAPHICS Plus, Version 4.

Multivariate analysis. Associations between benthic assemblages and acoustic regions were examined using multivariate statistical methods. Macrofauna data from the Hamon grab survey were divided into categories in order to determine the strength of association between these and acoustically distinct regions. These categories were: (1) all taxa excluding colonial species; (2) burrowing and infaunal species; (3) epifaunal species. Sample and species associations across the survey area for each of the three categories were assessed by non-metric multi-dimensional scaling (MDS) ordination using the Bray-Curtis similarity measure on 4th root transformed data (species categories 1 and 2) or presence/absence data (category 3) using the PRIMER software package (Clarke & Warwick, 1994). Low abundance species (i.e. with fewer than three individuals recorded through out the survey area) were removed from faunal category 1 in order to reduce the variability caused by these infrequently occurring species. Removing these species was also necessary to conform to certain limitations in the total number of species which can be used during certain tests within the PRIMER software (e.g. SIMPER—see later). The majority of species collected during the beam trawl survey were epifaunal species. Statistical analysis was therefore conducted on all taxa excluding colonial organisms using identical statistical methods as above on 4th root transformed data.

Analysis of similarities (ANOSIM, Clarke, 1993) was performed to test the significance of differences in macrofauna assemblage composition between samples. The nature of the groupings identified in the MDS ordinations were further explored by applying the similarity percentages program (SIMPER) to determine the contribution of individual species to the average dissimilarity between samples.

A correlation-based principal components analysis (PCA) was applied to ordinate results from the sediment particle size analysis. Prior to analysis, environmental variables were converted to approximate normality using a $\log(1+N)$ transformation. Analysis of similarities (ANOSIM, Clarke, 1993) was performed on particle size data to test the significance of differences in particle size composition between acoustic regions. The relationships between environmental variables and multivariate community structure were assessed using the BIO-ENV procedure within the PRIMER programme. In this procedure rank correlations ($\rho\omega$) between a similarity matrix derived from the biotic data and matrices derived from various subsets of environmental data are calculated, thereby defining suites of environmental variables which best explain the biotic structure. The RELATE programme was also applied to test for significant relationships between similarity matrices based on relative macrofauna abundances and measured environmental variables, and between categories of macrofauna collected during the grab survey. The Spearman rank correlation (ρ) was computed between corresponding elements of each pair of matrices, and the significance of the correlation determined using a permutation procedure.

Results

Acoustic data interpretation

Examination of the sidescan sonar data revealed the presence of eight acoustically distinct regions within the survey area (Figure 2). Underwater video footage established that differences between the acoustic regions were due to changes in substrate type, and that substrates were generally homogeneous in their distribution within each of the regions. Difficulties in identifying boundaries between acoustic regions in the north of the survey area (regions H, H/F, and F) were encountered due to the reduced sidescan sonar image quality caused by shallow water and surface noise. Boundaries between regions across the rest of the survey area were fairly distinct. Examples from the sidescan sonar record of the acoustically distinct regions, along with physical habitat descriptions derived from the underwater video footage, are illustrated in Figure 3.

Sediment characteristics and environmental variables

Examination of the grab samples on deck, and *in situ* study of the undisturbed sea-bed surface by the video camera attached to the side of the grab, confirmed that sediment characteristics within each acoustic region were relatively distinct and homogeneous in distribution. Results from the particle size analysis of grab samples, used in conjunction with information derived from the sidescan sonar mosaic and video footage, provided a clear understanding of the physical habitat characteristics within each acoustic region. A range of habitats were identified across the survey area: cobbles with attached algae in shallow inshore waters (depths around 10 m) (region H and H/F); areas of sand expressing different wave amplitudes (regions C and F); mixed coarse substrates (regions D and E); offshore gravelly sand with sand veneers (region B); offshore gravel and sand (>60 m) (region A).

An ordination by PCA of the particle size distributions from the grab samples is illustrated in Figure 4. There was a large degree of overlap between samples from most acoustic regions, and this was particularly apparent between regions with similar habitat traits (e.g. A and B, D and E, H and H/F), and reflects the subtle changes in sediment properties across the survey area. However, samples collected from within a number of the acoustic regions (A, C and H) tended to have similar particle size distributions, as depicted by the close proximity of replicate samples from these regions in the ordination (Figure 4). Samples on the left of the ordination (region H, E and D) tend to consist of coarser sediments (high percentage gravel) and are poorly sorted (high sorting coefficients). These parameters gradually reduce in magnitude across the ordination towards much finer (low percentage gravel), and well sorted (low sorting coefficient) sediments to the right of the ordination (Table 1). Table 2 shows the analysis of similarities results (ANOSIM, Clarke, 1993) for particle size data between samples. The high degree of overlap between regions and low degree of spatial clustering within the ordination is reflected in these results. Many of the regions were not statistically distinct in terms of their particle size distribution, and in general there was a high degree of particle size variability between replicate samples. Region C and H were the only two regions which tended to be statistically distinct from most of the other regions. Despite the fact that many of the regions were not statistically discrete in terms of their particle size distributions, the physical habitat characteristics (e.g. sea-bed morphology, degree of sediment stratification) were still distinct between these regions (Figure 3).

Biological data interpretation

A total of 233 taxa were identified from 43 Hamon grab samples collected from across the survey area. Univariate analysis revealed that regions A, D, E, and H had significantly higher numbers of macrofauna species, and that regions E and H had significantly higher numbers of individuals, than the remaining regions (Figure 5). Samples collected by beam trawl comprised of 113 taxa. Patterns in the number of species and individuals were similar to those from the Hamon grab survey. Regions A and D had the highest mean number of species, and regions A and H the highest mean number of individuals, although values from these regions were not always statistically higher than the other regions due to the high variability of these measures between replicate samples. Region C had the lowest mean number of species and individuals, and this was true for samples collected during both the beam trawl and grab surveys.

Figure 6 shows the output from non-metric multi-dimensional scaling (MDS) ordinations of data from both the Hamon grab and beam trawl surveys. Ordinations were carried out on the three macrofauna categories identified from the grab samples (following amalgamation of the 1–5 mm and >5 mm fractions) and on all species except colonial organisms from the beam trawl samples. Grouping of replicate samples from within acoustic regions is visible, which, following analysis of similarities (ANOSIM, Clarke, 1993), illustrates that in most cases there were significant differences in macrofaunal assemblage composition between acoustically distinct regions (Table 3). From the Hamon grab survey, regions D and E, D and H/F, and F and H/F were the only combinations of regions which did not have statistically distinct assemblages

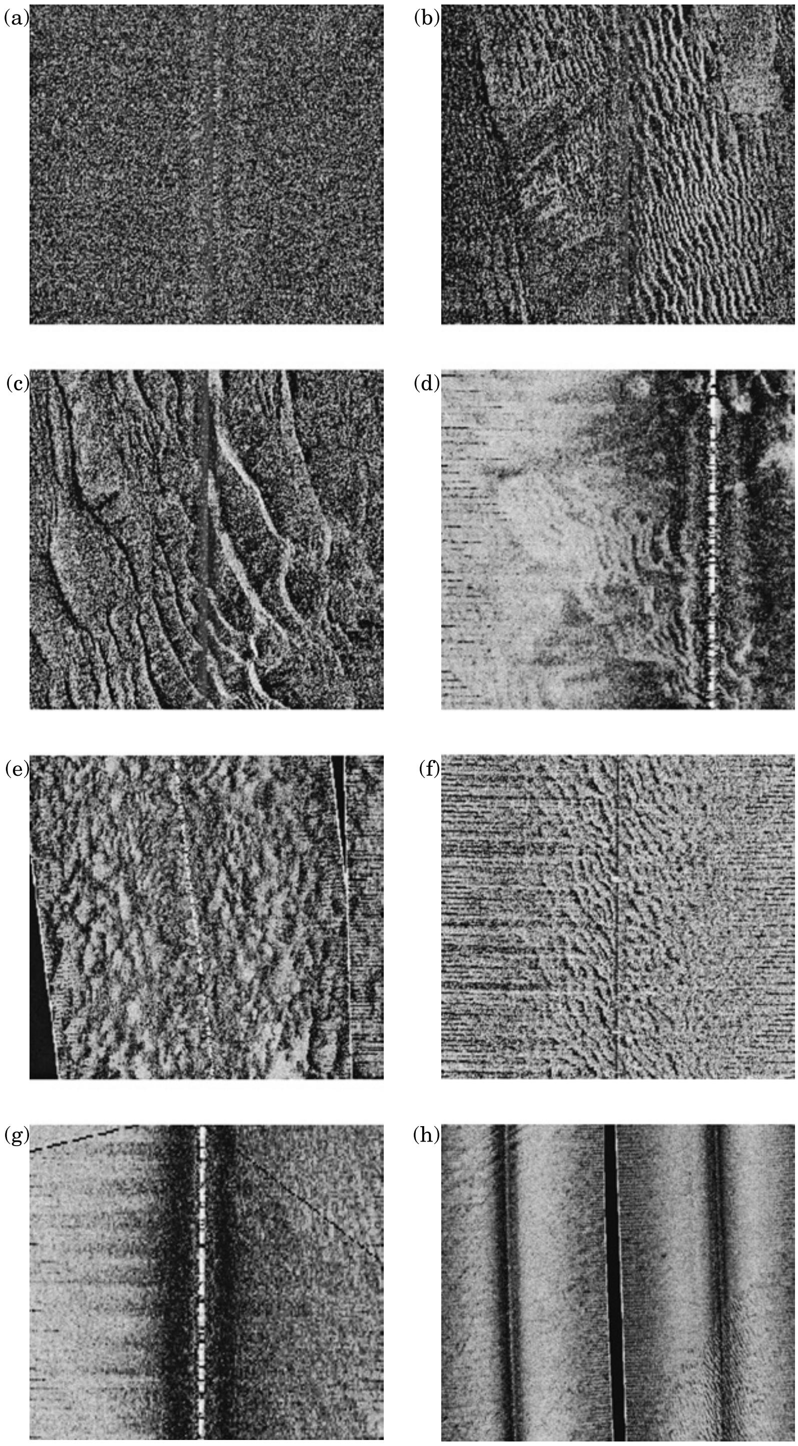

FIGURE 3. Examples of sidescan sonar images from the acoustically distinct regions. (a) Region A—Offshore sandy gravel; (b) Region B—Offshore sandy gravel with sand veneers; (c) Region C—Large sand waves; (d) Region D—Mixed heterogeneous sediment; (e) Region E—Uneven mixed heterogeneous substrates with boulders; (f) Region F—Inshore rippled sand; (g) Region H—Coarse gravel and cobbles with attached algae; (h) Region H/F—transition region between H and F, mixed substrates of cobbles and sand.

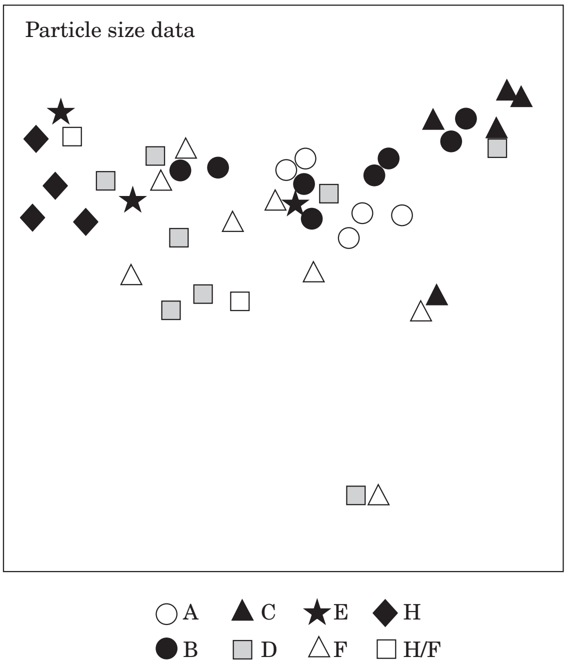

FIGURE 4. PCA ordination of particle size distributions.

for all three faunal categories. A number of other combinations of regions were not statistically distinct, but these combinations varied depending on which faunal category the ANOSIM test was applied to (Table 3). On the whole dissimilarity values between regions was generally high. High stress values for the ordinations are due to the high-dimensional data set and large number of samples included in the analysis. However, stress values between 0·1 and 0·2 still provide a valuable 2-D picture, and the ordinations offer a useful visual method of displaying the results (Clarke & Warwick, 1994).

The dissimilarity values between regions for the beam trawl data were much lower (Table 3). Although replicate samples are not strongly clustered in the ordination (Figure 6), the regions are spatially separated from one another with a low degree of overlap between regions, and these differences are supported by the ANOSIM results (Table 3). Due to the fact that only three replicate samples were collected from within each acoustically distinct region, it was only possible to achieve a significance level of 10% due to limitations in the statistical approach caused by the reduced number of permutations achievable between samples (Clarke, 1993). It should, however, be noted that this significance level can be used to infer an ecological difference in community structure between acoustic regions. In most cases assemblage structure was statistically distinct between acoustic regions at this significance level, with the exception of regions A and D, B and C, B and F, C and F and F and H.

Results from the RELATE analysis indicate a statistically significant similarity in biotic structure between all combinations of the three categories of benthic fauna from the grab survey ($P<0·05$). Similarly, there was a statistically significant similarity in the biotic structure between assemblages identified from the 2 m beam trawl survey and category 1 from the Hamon grab survey ($P<0·05$) (Table 4). These

TABLE 1. Particle size analysis data from Hamon grab samples collected from each acoustic region (means and standard deviation)

Acoustic region	% gravel	% sand	% silt/clay	Mean particle size (mm)	Sorting
A	28·83	66·22	4·94	0·86	2·30
N=5	(±11·71)	(±9·64)	(±2·45)	(±0·35)	(±0·14)
B	30·31	67·76	1·93	1·23	2·16
N=8	(±18·00)	(±18·57)	(±1·53)	(±0·97)	(±0·64)
C	4·37	92·33	3·30	0·52	1·12
N=5	(±6·68)	(±7·61)	(±6·24)	(±0·21)	(±0·64)
D	40·30	51·08	8·62	1·92	3·02
N=8	(±23·00)	(±21·55)	(±9·74)	(±1·63)	(±0·87)
E	61·19	35·41	3·40	3·68	2·64
N=3	(±19·33)	(±18·16)	(±1·94)	(±2·92)	(±0·45)
F	36·30	54·16	9·53	1·58	2·97
N=8	(±22·72)	(±15·90)	(±10·22)	(±1·24)	(±0·48)
H/F	55·62	38·20	6·18	3·22	3·10
N=2	(±22·40)	(±15·06)	(±7·33)	(±3·36)	(±0·22)
H	76·40	18·02	5·58	5·57	3·24
N=4	(±5·50)	(±4·38)	(±1·55)	(±1·15)	(±0·44)

TABLE 2. Analysis of similarity for particle size distributions of sediments between acoustically distinct regions

	A	B	C	D	E	F	H
B	n.s.						
C	0·008**	0·003**					
D	n.s.	0·076*	0·003**				
E	0·071*	n.s.	0·018**	n.s.			
F	n.s.	n.s.	0·001**	n.s.	n.s.		
H	0·008**	0·004**	0·008**	n.s.	0·057*	0·059*	
H/F	0·095*	n.s.	0·048**	n.s.	n.s.	n.s.	0·067*

n.s. Not significant; **significant at $P<0{\cdot}05$; *significant at $P<0{\cdot}1$).

Hamon grab	A	B	C	D	E	F	H	H/F	2 m Beam trawl	A	B	C	D	F	H
Mean No. Species	34	16	8	30	49	15	39	20	Mean No. Species	35	25	20	31	22	21
Mean No. Individuals	102	49	13	86	197	34	143	29	Mean No. Individuals	679	186	148	588	203	1384

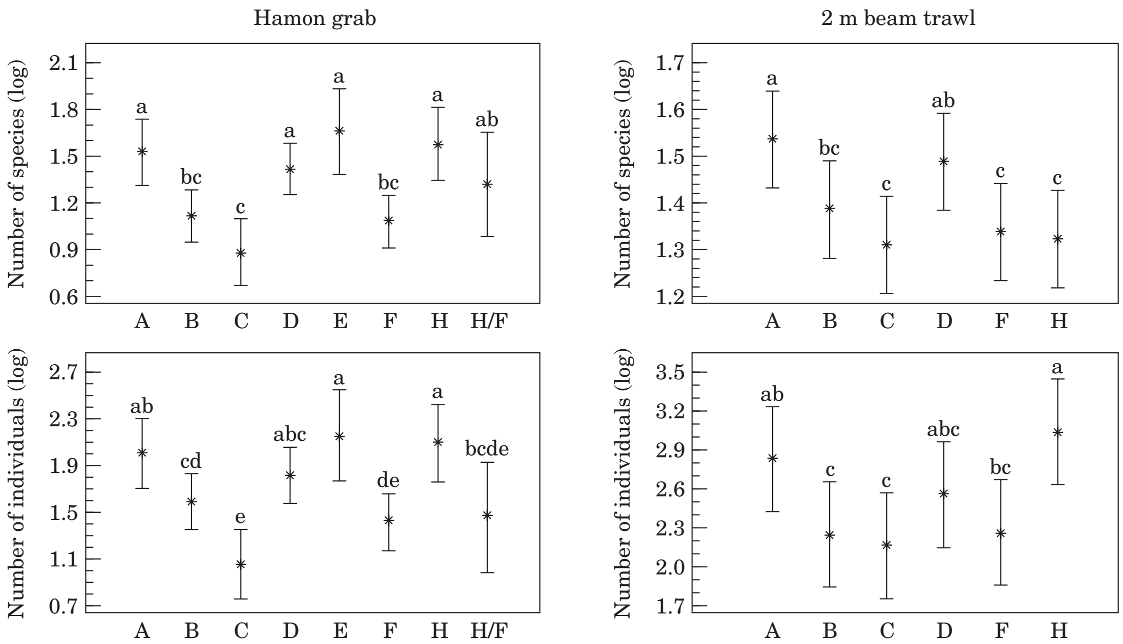

FIGURE 5. Summary of means and 95% pooled confidence intervals for the numbers of species and numbers of individuals from within each acoustic region. Values labelled with the same letter are not significantly different from one another at $P<0{\cdot}05$, following application of Fisher's LSD multiple comparison procedure.

results indicate that differences in the assemblage structure between acoustic regions were detectable, and that patterns in biotic structure were similar, irrespective of which fraction of the benthic community is sampled, or which sampling technique was used. Correlation between the environmental variables (psa data) and biotic matrices underlying the ordinations in Figures 4 and 6 were highly significant ($P<0{\cdot}01$) (Table 4).

BIO-ENV analyses were conducted on (dis)similarity matrices derived from the Hamon grab fauna categories 1–3 and environmental data to establish which suite of environmental variables best explain the biotic structure. Following analysis of data from all 43 stations (including only the particle size data within the environmental variables) percentage sand and sorting coefficient for faunal categories 1 and 2, and percentage gravel and sorting coefficient for

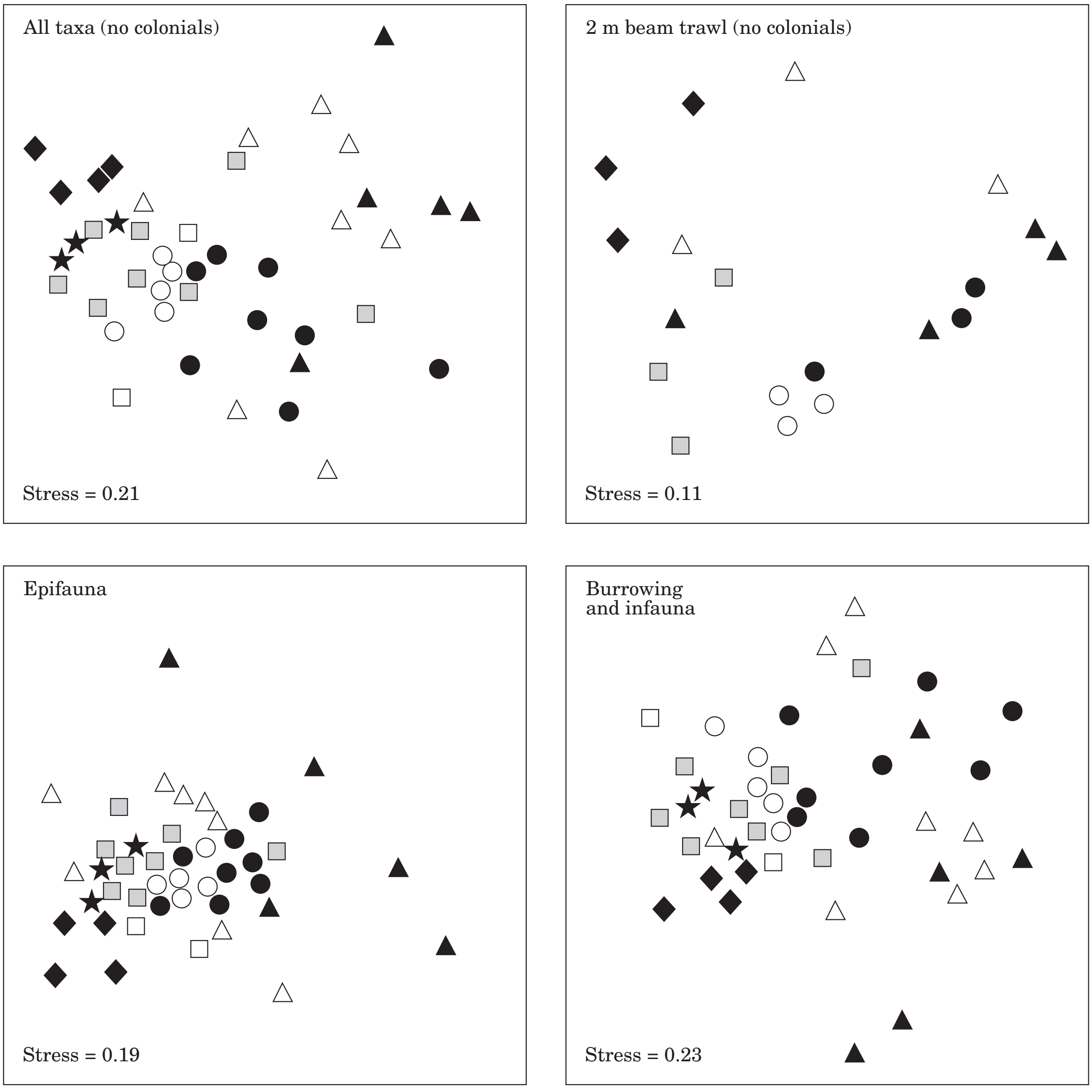

FIGURE 6. MDS plots for macrofaunal assemblages from the Beam trawl and Hamon grab surveys. Beam trawl data—4th root transformed. Hamon grab faunal category 1 (all taxa, no colonials) and faunal category 2 (burrowing and infauna)—4th root transformed, faunal category 3 (epifauna)—presence/absence transformation.

faunal category 3, were identified as the best combination of variables. In all cases however the weighted Spearman rank correlations between assemblage structure and environmental variables were low ($\rho\omega$<0·3). Tests were not carried out on beam trawl data and particle size data from the grab samples due to the fact that the trawl samples were collected over a stretch of sea-bed, and the particle size data was collected from single point stations.

Biotopes

Further exploration of the community groupings identified in the MDS ordinations, using the similarity percentages program SIMPER, was conducted for benthic data sets collected from the Hamon grab (using faunal category 1) and beam trawl surveys. Results revealed that the average similarity between replicate samples collected within an acoustic region

Table 3. Dissimilarities (%) between assemblages from within acoustically distinct regions based on 4th root transformed data (Hamon grab faunal categories 1 and 2, and 2-m beam trawl data) and presence/absence transformed data (Hamon grab faunal category 3)

	A	B	C	D	E	F	H
Category 1: All taxa—colonial and low abundance species removed (4th root transformed data)							
B	70						
C	87**	84**					
D	67	77**	90**				
E	66**	80**	92**	67			
F	83**	81**	83*	82**	85**		
H	74**	85**	94**	74*	63*	87**	
H/F	71**	77	89**	72	75*	82	81*
Category 2: Burrowing and infaunal species (4th root transformed data)							
B	73*						
C	86**	84**					
D	67	78**	90**				
E	66**	80**	91**	69			
F	82**	80**	83*	82**	84**		
H	74**	82**	92**	73	66	85**	
H/F	75**	82*	91**	75	76	84	79*
Category 3: Epifaunal species (presence/absence data)							
B	59*						
C	85**	84**					
D	64**	66**	88**				
E	63**	70**	91	56			
F	72*	70**	86**	68*	72		
H	74**	80**	94**	69**	64**	81**	
H/F	70**	71**	85	65	68*	69	69*
Beam trawl fauna—colonial species removed (4th root transformed data)							
B	49*						
C	61*	48					
D	50	63*	74*				
F	67*	62	62	64*			
H	67*	71*	79*	60*		64	

*Denotes significant difference at $P<0{\cdot}1$. **denotes significant difference at $P<0{\cdot}05$.

was relatively low, particularly for the Hamon grab data (Tables 5 and 6). This reflects the remaining large number of low frequency species (after removal of the lowest abundance species i.e. those species with fewer than three individuals throughout the survey area) within the data set that contribute to the high dissimilarity between replicates from within an acoustic region.

Table 4. Spearman rank correlations (ρ) between macrofaunal assemblage structure and environmental variables (particle size data), and between the faunal categories identified from the Hamon grab and 2 m beam trawl surveys

	PSA data	Epifauna	All taxa
Epifauna	0·408		
Burrowing and infauna	0·425	0·486	0·952
All taxa	0·475	0·600	
Beam trawl data	0·276		0·186

The output from SIMPER also indicates which taxa contribute the most towards the similarity between replicate samples from within each acoustic region (Tables 5 and 6). Characterizing species from each acoustic region identified from the Hamon grab survey were unsurprisingly very different from those identified from the beam trawl survey. Characterizing species identified from the beam trawl survey were typically larger and mobile epifaunal species. In contrast, those species identified from the Hamon grab survey tended to represent the smaller epifauna or infaunal members of the benthic assemblages. Independent of the type of sampling gear used, characterizing species from each acoustic region are typical for the substrate types present within the region. Several

TABLE 5. Results from SIMPER analysis of Hamon grab data (all taxa excluding colonial species, 4th root transformed), listing the main characterizing species from each acoustically distinct region. Average abundance, similarity percentage, and cumulative similarity percentage for each species and the overall average similarity between replicate samples from within each region are listed

Acoustic region		Average abundance	%	Cumulative %	Average similarity
A	*Echinocyamus pusillus*	12·6	8·0	8·0	
	Maldanidae	8·2	7·7	15·7	
	Ampelisca sp.	8·8	7·6	23·3	48·9%
	Aonides paucibranchiata	6·6	7·3	30·7	
	Lumbrineris gracilis	4·8	7·2	37·9	
B	*Echinocyamus pusillus*	12·3	20·9	20·9	
	Spisula sp.	4·5	12·5	33·4	28·6%
	Glycera sp.	1·5	11·9	45·3	
C	*Abra prismatica*	1·2	49·5	49·5	
	Glycera sp.	1·2	15·7	65·2	19·6%
	Praunus sp.	0·6	10·7	80·0	
D	*Lumbrineris gracilis*	4·6	15·4	15·4	
	Maldanidae	9·6	15·4	30·8	28·5%
	Amphipholis squamata	4·0	6·4	37·3	
	Echinocyamus pusillus	3·0	5·8	43·2	
E	*Ampelisca* sp.	5·0	7·2	7·2	
	Amphipholis squamata	13·0	7·2	14·4	
	Maldanidae	4·7	7·0	21·4	36·0%
	Lumbrineris gracilis	11·0	6·9	28·3	
	Sabellaria spinulosa	5·0	6·4	34·7	
F	*Ophelia borealis*	1·5	23·2	23·2	
	Bathyporeia sp.	4·4	19·2	42·4	21·9%
	Spisula sp.	2·6	10·7	53·2	
H	*Crepidula fornicata*	43·7	10·3	10·3	
	Scalibregma inflatum	7·2	7·0	17·3	44·2%
	Lumbrineris gracilis	4·2	6·8	24·2	
	Harmothoe sp.	3·5	6·7	30·9	

ubiquitous species were also identified ranging across all the regions, including *Pagurus bernhardus* and *Alcyonidium diaphanum*.

Using results from the SIMPER analysis of both Hamon grab and beam trawl data, along with information derived from the sidescan sonar mosaic and underwater video and photographic material, five discrete biotopes (physical habitats and associated biological assemblage) were identified:

Biotope A/B Echinoderm dominated (*Echinocyamus pusillus* and *Psammechinus miliaris*) gravelly sand with occasional sand veneers.

Regions A and B, whilst acoustically different, were very similar in terms of sediment characteristics and the benthic fauna. Particle size analysis revealed that both regions consisted of gravelly sands, with a high proportion of gravel on the sea-bed surface (determined form the video camera attached to the grab). Region B differed due to the presence of sand veneers over parts of the area, but the presence of these veneers did not appear to have a major influence on community structure. Both regions could not be statistically separated in terms of community structure (using faunal category 1 from the Hamon grab data), and were characterized by high numbers of the echinoderms *Echinocyamus pusillus* and *Psammechinus miliaris*.

Biotope C Clean mobile sand with *Abra prismatica*.

Region C was characterized by moderately large sand waves. Transport features suggested that the region was mobile and unstable, and this was reflected in the low number of species and densities within the area. The main characterizing species identified from the Hamon grab survey was *Abra prismatica* and, despite an average abundance of only 1·2 individuals, it accounted for 49·5% of the similarity between samples collected from this region. *Crangon allmani*, *Ophiura albida* and *Anapagurus laevis* were also identified as characterizing species from the beam trawl survey.

TABLE 6. Results from SIMPER analysis of beam trawl data (all taxa excluding colonial species, 4th root transformed), listing the main characterizing species from each acoustically distinct region. Average abundance, similarity percentage, and cumulative similarity percentage for each species and the overall average similarity between replicate samples from within each region are listed

Acoustic region		Average abundance	%	Cumulative %	Average similarity
A	*Psammechinus miliaris*	205	7·5	7·5	
	Aequipecten opercularis	102	6·6	14·1	
	Echinocyamus pusillus	62	5·7	19·8	79·4%
	Pagurus bernhardus	48	4·9	24·7	
	Ophiura albida	36	4·9	29·6	
B	*Pagurus bernhardus*	16	10·5	10·5	
	Crangon allmani	40	10·0	20·6	
	Anapagurus laevis	18	10·0	30·6	52·8%
	Ophiura albida	19	9·7	40·4	
	Pomatoschistus minutus	10	8·3	48·6	
C	*Crangon allmani*	19	12·0	12·0	
	Ophiura albida	20	11·1	23·1	56·6%
	Anapagurus laevis	17	10·7	33·8	
	Liocarcinus sp.	10	10·2	44·0	
D	*Anomia* sp.	120	7·9	7·9	
	Ocenebra	14	7·8	15·7	
	Crepidula fornicata	56	7·8	23·5	44·6%
	Psammechinus miliaris	18	7·7	31·2	
	Pagurus bernhardus	34	6·7	38·0	
F	*Pagurus bernhardus*	45	21·7	21·7	
	Pomatoschistus minutus	49	15·6	37·4	34·5%
	Macropodia sp.	3	10·7	48·1	
	Hinia	10	9·7	57·8	
H	*Crepidula fornicata*	1283	23·9	23·9	
	Ascidiella scabra	11	9·1	33·1	60·2%
	Pagurus bernhardus	15	8·8	41·9	
	Macropodia sp.	10	8·2	50·1	

Biotope D/E Polychaete-dominated mixed, heterogeneous sediments.

Both regions D and E consisted of very mixed, heterogeneous sediments and, although these regions appeared very different acoustically (Figure 3), they supported similar benthic communities. Both regions contained a large percentage of coarse sediments, and whilst the surface topography appeared very different between regions, particle size distributions were similar. Both regions had very high numbers of species and individuals, and were dominated by polychaetes such as *Lumbrineris gracilis* and Maldanid species, as well as a number of molluscan species (see Tables 5 and 6). Particle size distributions were similar to those in regions A and B, but with a higher percentage of coarse material, and there were common, characterizing species between all four of these regions (e.g. *Echinocyamus pusillus*, *Psammechinus miliaris*, *L. gracilis*). However, differences in habitat and community structure were great enough to distinguish between biotope A/B and D/E.

Biotope F Sand and gravelly sand with *Ophelia borealis*, *Bathyporeia* sp. and *Pomatoschistus minutus*.

The sea-bed surface within region F was predominantly rippled sand, which was clearly identified from the acoustic record and underwater video/photography. Particle size analysis revealed that the region contained a higher percentage of coarse material than initially expected and, as a result, the particle size distribution of sediments within this region was not statistically distinct from most other regions. However, the surface material appeared to be predominantly sandy, and this was reflected in the characterizing fauna, *Ophelia borealis*, *Bathyporeia* sp. and *Pomatoschistus minutus*, all of which prefer sandy substrates.

Biotope H Cobbles with algae (unidentified), and *Crepidula fornicata*.

Underwater video revealed that region H was very distinct from other regions. The substrate within the region was very coarse, consisting of a high percentage of cobbles and gravel supporting a large number of epifauna and flora (algal species were abundant within the region but were not identified or quantified). The region supported very high numbers of *Crepidula fornicata*, which was identified as the main characterizing species from both the beam trawl and Hamon grab surveys. Other characterizing species included *Scalibregma inflatum*, *Lumbrineris gracilis*, and *Ascidiella scabra*.

Region H/F did not appear to be a distinct region. Problems were encountered identifying the boundaries of the region from the sidescan sonar record, and the region appeared to form a transition between regions H and F. For this reason the area has not been identified as a separate biotope, and has been treated as a zone of transition between the two neighbouring regions.

Discussion

In the English Channel, a number of studies have attempted to identify and explain distribution trends in benthic species, assemblages and habitats (Holme, 1961, 1966; Cabioch, 1968; Davoult *et al*., 1988; Sanvicente-Anorve *et al*., 1996). Community types identified in these studies, along with those identified from other regions (e.g. North Sea: Dyer *et al*., 1983; Basford *et al*., 1989, 1990; Eleftheriou & Basford, 1989; Kunitzer *et al*., 1992; Rees *et al*., 1999) show some parallels with the five biotopes identified in the current study. However, most of these earlier studies were conducted over large areas (i.e. whole sea areas) where differences in the biogeographical ranges of species might be expected to contribute to changes in community structure. The current study identified biotopes over a relatively small area of sea-bed, at a much higher resolution, where biogeographical constraints on distributions of species would clearly have no influence on community structure. Instead, more localized variables were responsible for changes in species distributions, such as sediment granulometry (Table 4). It should therefore be recognized that the five biotopes identified in the current study may represent sub-sets within the community types proposed in the past, due to the differences in scale and sampling intensity between the current and past studies.

Glémarec (1973) divided communities on the bases of thermal stability, recognizing that certain species have limited temperature tolerances, which in turn influences the species composition of a community within a defined geographical area. He proposed three 'étages ' based on this principle, but recognized the importance of sediment granulometry in determining community structure at higher resolutions within each étage. The five biotopes identified in the current study all fall within Glémarec's coastal étage. Within this étage, Glémarec (1973) proposed a coarse sand community which he claimed had a number of subtly different forms, or facies, depending on the ratio of gravel to coarse sand within the area. Many of the biotopes identified in the current study appear to fall within this class.

Parallels between biotope F and Glémarec's *Ophelia borealis–Travisia forbesi* facies inhabiting medium sands are easy to draw, as both have sediment characteristics (medium sands) and fauna (*O. borealis*) in common. This association is widely recognized in the eastern Channel (Davoult & Richard, 1988; Dewarumez *et al*., 1992; Sanvincente-Anorve *et al*., 1996) and it is likely that biotope F falls into this category. Glémarec (1973) also identified a subtle variation in this community associated with an increase in the percentage of finer sand and the appearance of *Abra prismatica*. This subtle change in granulometry and discriminating fauna is evident within acoustic region C, allowing this region to be identified as a distinct biotope. This biotope also shows a number of similarities to the boreal offshore-sand association described by Holme (1966), and to the fine sand association of the northern North Sea identified by Eleftheriou and Basford (1989), both of which were characterized by fine sands and the presence of *A. prismatica*.

Biotopes A/B and D/E contained a large number of common species, despite the fact that the regions could be distinguished statistically. Sediment granulometry was similar between the regions, and the physical habitat of the two biotopes could only be distinguished through the use of high-resolution sidescan sonar and underwater video techniques, which revealed subtle differences in the physical structure (e.g. surface morphology) of the habitats. It is possible that these differences would have been overlooked if the survey had only employed traditional sampling methods (i.e. grabs), or if the survey had been conducted at a broader scale. It is likely that these two biotopes represent facies of the same community, and it is therefore unlikely that parallels with other communities described in the past will be identified from the literature for both biotopes. There are, however, certain similarities between biotopes A/B and D/E and the boreal offshore gravel association described by Holme (1966), namely the high numbers of the green sea urchin *Echinocyamus pusillus* on coarse, gravelly sediments. Davoult and Richard (1988) and Rees *et al*. (1999) also identify distinct faunal groups with high numbers of *E. pusillus* associated with coarse substrates, similar to biotopes A/B and D/E. It is plausible, therefore, that these two biotopes represent facies of the coarse sand community within the coastal étage proposed by Glémarec (1973).

Multivariate analysis of biological and sediment granulometry data from region H revealed that this region was statistically discrete from most other regions, and was characterized by the presence of coarse deposits, and high abundances of *Crepidula fornicata* and attached macrophytes. References to similar associations in previous studies are rare. Holme (1961) identified the presence of *C. fornicata* in the central regions of the channel, and there may be parallels with the pebbles community identified in a number of studies (Davoult & Clabaut, 1988; Dewarumez *et al.*, 1992; Migné & Davoult, 1997), although there are few characterizing species in common.

The concept of discrete communities versus continua has long been debated. Glémarec (1973), after reviewing the arguments in a number of earlier studies, concluded that there are no sharp distinctions between neighbouring communities but rather gradual changes in the composition of the fauna without discontinuities. However, he recognized that in order to produce maps it is necessary to draw demarcation lines, and as a result communities are defined which relate to the peaks of frequency (or noda) within the continuous gradient of faunal composition. Basford *et al.* (1989, 1990) also reported that ' community types ', identified from surveys in the North Sea, were found to grade into one another along continuous environmental gradients, even though discrete assemblages could be identified statistically and characterized by particular species. These studies were conducted at very broad scales, where gradients responsible for the changes in assemblage structure (temperature, salinity and depth), although often gradual, were nevertheless significant over the entirety of the survey area.

Other factors, such as sediment characteristics, are thought to have a greater influence on assemblage structure at more localized scales (Holme, 1961, 1966; Glémarec, 1973; Eleftheriou & Basford, 1989; Seiderer & Newell, 1999). Substratum types can often show discontinuities across a region which may give rise to distinct boundaries between neighbouring assemblages. The use of sidescan sonar in the current study enabled such boundaries to be identified and mapped. Designing subsequent biological surveys around the acoustically distinct regions determined from the sidescan sonar data made it possible to test whether discrete assemblages existed within these boundaries. In some cases, discrete physical boundaries could be identified between two neighbouring regions which did not support discrete assemblages (e.g. A and B; D and E). In these cases differences between the adjacent regions were too subtle to have a significant effect on the composition of the benthic community. However, discrete benthic assemblages did appear to be contained within the boundaries of most other regions (region C, F and H), suggesting that at this scale faunistic boundaries can exist, but may only be recognized using appropriate techniques (e.g. high resolution sidescan sonar).

Even though faunistic boundaries were identified within parts of the survey area, there was also evidence of spatial gradation of habitats and communities from one area to the next. This was particularly apparent in the north of the survey area between regions H, H/F and F. It is possible that boundaries between these regions did exist, but that the poor sidescan sonar record from this area caused by the shallow water prevented them from being identified. However, evidence from the underwater video footage did suggest the presence of an east-west sediment gradient from sandy substrates in the east to coarse gravel and cobbles in the west. This gradual east-west sediment transition was also reflected in the biological data; regions H and F were characterized by statistically different benthic communities, with region H/F comprising common species from both these regions, thus forming a non-statistically distinct transition region. Such transition regions between distinct habitats/assemblages, sometimes referred to as ecotones, have been described in the past (Dewarumez *et al.*, 1992), and it is arguable whether or not they should be treated as entities. The evidence from the current study suggests that the presence of either discrete habitat/faunistic boundaries, or of sediment/faunistic gradients between statistically distinct, adjacent regions is a site-specific phenomenon. However, the lack of clearly definable boundaries between adjacent habitats can cause major problems when attempting to produce high-resolution sea-bed maps due to difficulties in determining where demarcation lines should be drawn.

When classifying biotopes the main characterizing species of an area are listed along side a description of the physical habitat (Connor *et al.*, 1997; Davies & Moss, 1998). The combined use of sidescan sonar and underwater video/photographic techniques proved to be a useful approach in order to provide information concerning the physical characteristics of an area of sea-bed. However, when characterizing the sea-bed assemblages of an area the type of sampling gear used has a profound effect on how the community is described. In the current study a 0·1 m^2 Hamon grab and a 2 m beam trawl were used to characterize the benthos within the acoustic regions. It is evident from the characterizing species listed in Tables 5 and 6 that the use of either of these techniques in isolation would result in the derivation of different biotope descriptions. This is due to differences in the nature of the sample gear which results in the collection of a different fraction of the benthic community. Therefore the type of sampling gear has a considerable bearing not only on the identification of characterizing species, but also on the power to discriminate between habitat types on the basis of biological traits. The relevance of the characterizing species for the management of activities within a mapped region is another important practical consideration which bears upon the biological sampling techniques employed. For this reason the deployment of a combination of sampling techniques would provide a more realistic means of describing the benthic ecosystem, accepting that the capacity to discriminate between habitat types on biological grounds may often be method dependent (Holme, 1961; Rees *et al.*, 1999).

The methodology developed during this study, using a combination of acoustic, biological and underwater video/photographic sampling techniques, proved to be successful in mapping sea-bed biotopes within the survey area. However, small-scale sediment variability within an area could pose problems when using the current approach to sea-bed mapping. The interpretative process used to divide the sidescan sonar mosaic into acoustically distinct regions will be more difficult to conduct in areas where the sea-bed comprises complex, heterogeneous substrates, where boundaries between different habitats are indistinct. The scale at which acoustically distinct regions are defined is an important issue, and has profound implications for the design and effort required for subsequent biological surveys. The presence of relatively strong physical gradients across the survey area in the current study, with regions displaying a relatively high level of sediment homogeneity, undoubtedly facilitated the interpretation process and, in most cases, allowed discrete habitats and assemblages to be identified. The implications of small-scale variability of sediments for the mapping of sea-bed assemblages has yet to be investigated, but even taking these potential limitations into account, the current method of using a combination of acoustic and biological survey techniques still appears to hold many advantages over more traditional approaches.

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